Declines and Conservation of Himalayan Galliformes

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Abstract

The Greater Himalaya has been identified as a key conservation region that supports high levels of biodiversity but has exceptionally high proportions of threatened species. One taxonomic group that is thought to be of particular concern is the bird order Galliformes. The Greater Himalaya is home to 24 species of resident Galliformes with a variety of ecological characteristics, geographical distribution patterns and abundance levels. Our current knowledge of South Asian Galliformes and Himalayan species in particular, contains many gaps. For example, it is suspected that many Himalayan Galliformes have undergone marked population declines but as to what extent they have declined and even the current status of some species is not fully known. There is a similar paucity of knowledge regarding both the distributions of the rarest of Himalayan Galliformes species and how well the current protected area network represents such distributions.

Here I provide new insights into the distribution of the rarest Himalayan Galliform, the Critically Endangered Himalayan Quail (*Orphrysia superciliosa*) by using two proxy species with similar habitat preferences to create an environmental niche model. I show that by calculating an estimate of extinction likelihood, we have good reason to believe that the Himalayan quail to be extant and that recent searches in Nepal would be better targeted in North East India.

Moving from single species to multiple species, I then examine long-term population changes across all Himalayan Galliformes by using changes in geographic range size as a proxy. I show that population changes for this suite of species both within and outside the Himalaya can help us to set conservation priorities and baselines. In addition, it can help us to identify species that have undergone large population changes that are not reflected in contemporary IUCN Red List statues.

Species with small geographic ranges are currently top priorities for conservation efforts because they are thought to be at a greater risk of extinction. However, because it is also easier to track long term population changes over smaller spatial scales, concern exists that we may have underestimated the declines and therefore the extinction risk of more

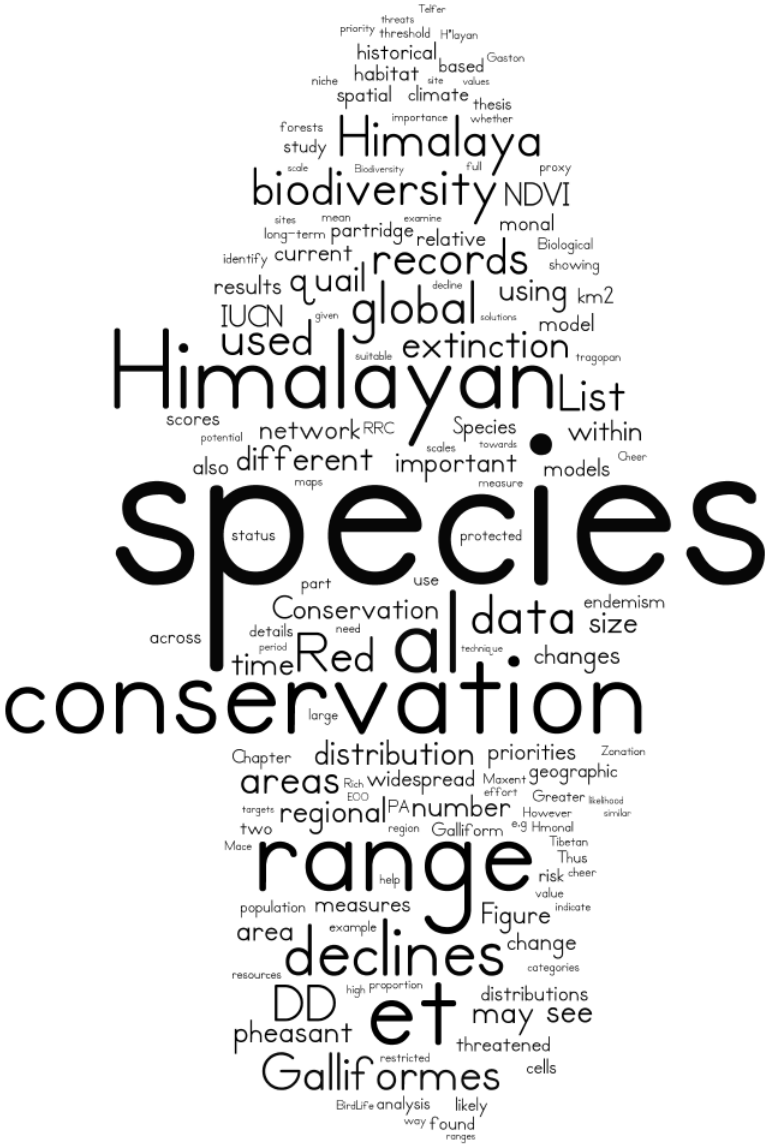
widespread species. I show that across the entire Galliformes taxon, geographic range size does not predict the rate of geographic range decline.

Finally, I move from population declines across all Galliformes to distributions of Himalayan Galliformes and assess how well the current protected area network represents such species. Using a combination of species distribution modelling and spatial prioritisation software, I show that the current protected area network in the greater Himalayas could be improved to offer better coverage for Himalayan Galliformes.

I conclude by discussing the generality of my results and how they can be applied to other taxa and localities. Finally I make a series of recommendations for future Galliformes research and conservation within the Himalaya.

Dedication

This thesis is dedicated to Granny and Grampy Dunn. RIP.



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The following people made specific contributions to my thesis, which are outlined below: Graeme M. Buchanan, RSPB (Chapters 2 and 5) provided species distribution modelling advice and read through a draft of my chapter two manuscript; Richard J. Cuthbert, RSPB (Chapter 2) provided field insights

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Everything else was written and analysed by Jonathon Dunn. Mark Whittingham and Philip McGowan oversaw and directed the research and read drafts of the thesis chapters.

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Chapter 1. Introduction

1.1 Global biodiversity in crisis

Global biodiversity – that is, the variability within and between both living things and ecosystems (Secretariat of the Convention on Biological Diversity, 1992) – is in a state of crisis: we are currently losing species faster than any other time in Earth's recent history (Millennium Ecosystem Assessment, 2005). Losing species through extinctions is normal, as approximately 99% of life that ever existed is now extinct (Novacek, 2001), but according to the latest estimates the current rate of loss is in excess of the usual, background rate by about one thousand-fold (Pimm et al., 2014). This level of extinction is sufficiently high that comparisons have been made to the five mass extinction events that have occurred throughout the last 3.5 billion years of geological time where life has existed (e.g. Lawton and May, 1995; Wake and Vredenburg, 2008). Arguably, the most well-known of these mass extinction events happened approximately 65 million years ago at the end of the Cretaceous period and saw the extinction of the dinosaurs (e.g. Alvarez et al., 1980; Macleod et al., 1997; Archibald et al., 2010; Brusatte et al., 2014). The underlying cause of these historical mass extinction events is still debated: hypotheses range from the impact of asteroids (Schulte et al., 2010) to global sea level rise (Hallam and Wignall, 1999), but there does not appear to be one single cause that can be consistently attributed to having driven these extinctions (Arens and West, 2008). What is clear is that in contrast to previous mass extinction events, the current global biodiversity crisis is due to the influence of humans on global ecosystems (Balmford et al., 2003). Humanity can and has influenced nature in many ways, but at its simplest, humans compete with other species for resources and space. This competition has manifested itself most clearly through the conversion of natural environments (Pimm and Raven, 2000), aided and abetted by the other principle threats of invasive species, pollution, human overpopulation and over-harvesting (Groombridge and Jenkins, 2002; Wilson, 2002), all of which are implicated in driving biodiversity loss (e.g. Butchart et al., 2005; Secretariat of the Convention on Biological Diversity, 2010). These threats are not mutually exclusive and can interact synergistically with each other in different and often unexpected ways (Brook et al., 2008). For example, a consequence of

deforestation is often habitat fragmentation and the creation of edges, which in turn have been shown to increase the susceptibility of a habitat to invasive species (Harper et al., 2005). As a result, anthropogenic threats have led to a fundamental change to global biodiversity including the structure of habitats and ecosystem functioning (Vitousek, 1997; Millennium Ecosystem Assessment, 2005).

Without mitigation, these drivers of species extinction are unlikely to abate in the near future. Biodiversity loss has been linked to human population density (McKee et al., 2004; Luck, 2007) and projections from the United Nations suggest that the global human population (7.2 billion as of 2013) is set to increase further by 1 billion over the next 12 years and is likely to reach 9.6 billion by 2050 (United Nations Department of Economic and Social Affairs Population Division, 2013). This is especially troubling as some of the areas with the most species are also the areas where human population growth is likely to be fastest in the future (Bawa and Dayanandan, 1997; Sodhi et al., 2004), suggesting further conflicts between people and biodiversity.

Furthermore, in the future novel threats such as anthropogenic climate change are likely to be of increasing importance in addition to the established threats outlined. As the worldwide human population has grown, increasing demands were placed on natural resources and in particular, increasing energy consumption. This drove the increased burning of fossil fuels, which led to increased greenhouse gas emissions (IPCC, 2013). As a consequence, the earth has warmed by 0.74°C over the last 100 years (IPCC, 2012) and if greenhouse gas emissions do not abate, could rise by a further 0.3-4.8°C within the 21st century depending on the scenario (Stocker et al., 2013). Climate change is likely to have large effects on biodiversity in its own right, e.g. changes to phenology (Parmesan and Yohe, 2003; Root et al., 2003), but may interact with other existing threats. For example, global warming has been linked to the spread of the chytrid fungus (Pounds et al., 2006), which has had a catastrophic effect on the world's amphibians (Berger et al., 1998; Olson et al., 2013). It has been pointed out that the current coupling of global species declines with rapid, unusual climate change and high atmospheric CO_2 levels bears much resemblance to the conditions of historical mass extinctions (Barnosky et al., 2011).

The consequences of biodiversity loss are likely to be profound, both for biodiversity itself and humans who depend on biodiversity. It is only recently that there has been a deeper consideration of just how extensively humanity is reliant on biodiversity and the functioning of natural ecosystems (Millennium Ecosystem Assessment, 2005; TEEB, 2010), the latter has been termed 'ecosystem services' (Daily, 1997). These services are broad and range from

those that involve the production of renewable resources such as food (Nabhan and Buchmann, 1997), to those that lessen environmental change such as flood mitigation (Bradshaw et al., 2007) and those that are related to human culture, values and well-being (Clark et al., 2014). Attempts have been made to quantify financially the value of these ecosystems services to our global economy, although placing an exact numerical value on the benefits of wild nature to humanity is difficult and in some senses is impossible (Nunes and van den Bergh, 2001). One early study came up with the annual value of $38 trillion (Costanza et al., 1997), but these figures have been heavily criticised with later estimates now between $4.4 trillion and $5.2 trillion a year (Balmford et al., 2002). Despite these accounting difficulties one fact is indisputable: if natural capital were completely lost, it would wreak havoc on the global economy and all aspects of human society.

1.2 How to halt and reduce biodiversity loss?

Biodiversity is being lost at an alarming rate due to anthropogenic influences and the effects are likely to be far-reaching. Ameliorating or lessening the impact of these influences to preserve biodiversity for the future is one of the biggest challenges human society currently faces (e.g. Cardinale et al., 2012). This reduction of threats is part of biodiversity conservation, which in its broadest sense is concerned with protecting multiple aspects of biodiversity from extinction (Soulé, 1985).

The increasing recognition of the importance of the benefits that biodiversity provides to humanity has been reflected in international law. In 2002 a commitment was made by world leaders at the Convention on Biological Diversity (CBD) to 'achieve by 2010 a significant reduction of the current rate of biodiversity loss' (Convention on Biological Diversity, 2009). These ambitious targets were not achieved (Butchart et al., 2010), which led the CBD member states to make further commitments to a new Strategic Plan for Biodiversity for the years 2011-2020 (designated by the UN as the 'United Nations Decade on Biodiversity'), which included 20 Aichi Biodiversity Targets that contribute to 5 strategic goals (CBD, 2010). These 20 targets all build upon the earlier 2010 targets and specifically aim to address some of reasons why the earlier targets failed (Butchart et al., 2010) through a more comprehensive managing and monitoring framework.

So why were the 2010 CBD targets not met? In theory achieving those targets was feasible. First, conservation itself has been shown to be effective at halting biodiversity loss, reducing the number of species that would have declined and become extinct (Hoffmann et al., 2010). Second, it has been shown that it makes

financial sense to conserve biodiversity: for example, it has been calculated that the costs required by conservation to protect ecosystem services are minimal in requirement to the value of the services themselves: one estimate suggests that for just $50 billion a year we could protect services worth $5 trillion a year (Balmford et al., 2002) and that the total costs of biodiversity conservation are less than 20% of the annual global consumer spending on soft drinks (McCarthy et al., 2012). Nevertheless, almost all parties to the CBD blamed in part, a lack of financial investment that doomed the 2010 targets to failure (Convention on Biological Diversity, 2009; Waldron et al., 2013).

At first, this lack of financial investment in biodiversity conservation is puzzling given that conservation has been shown to lessen biodiversity loss and is affordable. Part of the problem is a disagreement between governments over how a reduction in biodiversity loss should be implemented and specifically, who should cover the costs of conservation (Balmford and Whitten, 2003). This disagreement is related to the fact that biodiversity, the human population, human wealth and associated anthropogenic threats are not distributed across the globe equally (Cincotta et al., 2000; Gaston, 2000). Thus, the countries with the greatest financial resources are rarely those with the greatest amounts of biodiversity (Myers et al., 2000). This means that countries that stand to make the highest conservation gains through investing in natural capital often cannot afford to do so (McKinney, 2002; Balmford and Whitten, 2003). This is further compounded by the fact that most conservation is parochial, in the sense that people tend to care only about the biodiversity that is near to where they live (Hunter and Hutchinson, 1994) and that it can be difficult for people to see the long-term economic benefits that biodiversity brings (Tisdell, 2011). Although the deadline for the 2010 targets has now passed, a lack of financial investment in biodiversity is a problem that still seems to persist: a recent study found that our current investments in natural capital are woefully inadequate if we wish to achieve the 2020 targets (McCarthy et al., 2012).

The second cause for the failure of the 2010 targets relates to a lack of political will (Convention on Biological Diversity, 2009). One of the main reasons for this is a perceived competition between reducing biodiversity loss and reducing human poverty (Sanderson and Redford, 2003; Adams et al., 2004; Griggs et al., 2013), the latter of which takes place through food production, human habitation and resource extraction (Abbitt et al., 2000; James et al., 2001; Balmford et al., 2004; Naidoo and Iwamura, 2007). Another confounding factor is poor governance, which can reduce the success of conservation projects (Smith et al., 2003). It is only recently that some of these social and economic issues are being better considered by conservationists (Carpenter

et al., 2006). Nevertheless, without more cooperation and investment between countries (Waldron et al., 2013), increased global leadership and a better consideration of both social and economic issues (Polasky, 2008), implementing biodiversity conservation measures across a global stage is likely to remain challenging.

Given this background, although it is theoretically affordable to protect biodiversity and ecosystem services through conservation, a lack of investment in conservation resources and a lack of political will means that it is currently improbable that we will alleviate all anthropogenic threats and halt and reverse biodiversity loss in all of its manifestations. As such, the central challenge is how to act – how should we conserve biodiversity in the face of a funding shortfall if we are to achieve our 2020 commitments?

Answering this question is potentially very complex. To help simplify things, a closer examination of the Aichi 2020 targets is needed. Rather than examining all 5 strategic goals and 20 targets in detail, I will focus on just one goal: Aichi strategic goal C, which aims to 'improve the status of biodiversity by safeguarding ecosystems, species and genetic diversity' (CBD, 2010). Targets 11-13 are contained within this goal but I focus on issues relating to Targets 11 and 12 (see Table 1.1) only and specifically, examine species rather than ecosystems or genetic diversity (although I do briefly consider phylogenetic diversity in Chapter 5). These two targets make clear that in order to improve the status of biodiversity we need to consider two things: both species themselves and the areas where they live.

Aichi targets 11 and 12 demonstrate that not all species and not all areas are considered equal, i.e. that there is a variation in the identity of species that are at risk of extinction and there is a variation in the importance of areas for biodiversity. Thus, implicit in these two targets is the concept of prioritisation – that the limited resources available to conservation should be directed where it will be most effective. There are multiple approaches and techniques used in conservation prioritisation (e.g. Smith and Theberge, 1987; Costello and Polasky, 2004; Brooks et al., 2006; Sarkar et al., 2006; Wilson et al., 2009), which is an extremely dynamic topic for research. In this thesis, I mainly consider priority setting approaches that focus on species at risk from extinction and areas that are important for biodiversity. Effective conservation priority-setting of both species and sites is therefore a key component of solving this central problem of how to conserve global biodiversity and these two topics are outlined in greater detail the following sections.

Aichi target number	Description
11	'By 2020, at least 17 per cent of terrestrial and inland water areas, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes.'
12	'By 2020 the extinction of known threatened species has been prevented and their conservation status, particularly of those most in decline, has been improved and sustained.'

Table 1.1: Aichi 2012 targets 11 and 12 (CBD, 2010).

1.3 Which species to conserve?

Species are sometimes called the natural units of biodiversity (Agapow et al., 2004). The extinction of a species is nearly always irreversible, which makes it something to be avoided as much as possible (Erickson, 2000; Sandler, 2014). There are numerous political, social, emotional, ethical and aesthetic criteria that may be considered when decided which species should be conserved (De Grammont and Cuarón, 2006; Miller et al., 2006), but in the context of this thesis it is helpful to introduce and focus on the concepts of vulnerability and irreplaceability as they relate to species level prioritisation. Brooks et al. (2006) discuss this topic in further detail, but in essence, vulnerable species are those that are most at risk of immediate extinction and irreplaceable species are those that are either endemic (with small geographic ranges) or taxonomically unique. In this thesis, I give greater attention to conservation approaches that examine aspects of vulnerability as opposed to irreplaceability. Thus, the prioritisation of resources to minimise species extinctions is both a key concern of conservation biology (e.g. Vane-Wright et al., 1991; Ricketts et al., 2005) and Aichi Target 12 (see Table 1.1). Implementing this prioritisation of resources to minimise extinctions in practise, however, is not always straightforward.

Traditionally, conservationists have emphasised the need to direct resources to the species that most urgently require conservation now at whatever the cost (Parr et al., 2009; Wilson et al., 2011). Central to this approach is the IUCN Red List. The Red List procedure is the pre-eminent way of categorising a species' global extinction risk against a set of pre-defined criteria (IUCN, 2012). There are five IUCN criteria that are used to assess a species' extinction risk. Broadly, these criteria focus on measurements of local rarity (i.e. a small

population or geographic range) and species declines (i.e. a sharp reduction in population or geographic range). Once assessed, species are partitioned into one of five categories that indicate the extinction probability of an extant species that range from 'Least Concern' to 'Critically Endangered'. Species listed as 'Critically Endangered', 'Endangered' or 'Vulnerable' are commonly referred to as threatened species or those with a high vulnerability to extinction and are considered top conservation priorities (Possingham et al., 2002). Thus, in this way the IUCN Red List category of a species often acts as a basis for species-specific priority-setting, although the Red List was not specifically designed for this purpose (Mace and Lande, 1991; Lamoreux et al., 2003; Rodrigues et al., 2006).

Increasingly it has been recognised that prioritising conservation attention towards species most threatened with imminent extinctions paradoxically may not always be the best way of minimising the number of species extinctions (Bottrill et al., 2008). This is because other factors such as the cost of conservation action (e.g. Naidoo et al., 2006) and the likelihood conservation intervention will achieve the desired action may need to be considered (Wilson et al., 2009). For example, some highly threatened species may require continued investment indefinitely just to postpone extinctions that may happen anyway. Assessing the likelihood that conservation intervention will succeed (for a discussion of this see Redford et al., 2011) or knowing how much conservation intervention is needed (e.g. Boyd et al., 2008) is difficult. Nevertheless, continued spending of scarce resources on species that face imminent extinction (such as critically endangered species) may fail to minimise extinctions in the long run (Bottrill et al., 2008) or even divert resources from species that could better respond to conservation spending, such as more widely distributed and abundant species, which are typically thought to have a low vulnerability to extinction. This focus on cost-efficiency may result in the seemingly paradoxical decision not to invest in some highly threatened species (termed 'conservation triage'; Bottrill et al., 2008). Although it is distasteful to some who view this as a defeatist strategy (Mittermeier et al., 1998; Pimm, 2000; Marris, 2007; Jachowski and Kesler, 2009; Parr et al., 2009), conservation triage is an important aspect of conservation. This is implicitly recognised in Aichi target 12 (see Table 1.1), which aims not only to prevent extinctions in the short-term, but to improve and sustain the conservation status of known threatened species. Thus, proponents of conservation triage argue that in order to improve and sustain the conservation status for as many threatened species as possible, some extinctions are inevitable.

These two different but non-mutually exclusive approaches are sometimes characterised as 'short-term firefighting' versus 'preventive conservation' (or

reactive vs. proactive conservation; Brooks et al., 2006; Wilson et al., 2011) and in this thesis I examine elements of both (see section 1.8). Throughout, I make use one of the IUCN Red List's five criteria to examine different conservation strategies relating to threatened species. Specifically, I examine aspects of species declines (as mentioned in Aichi target 12) and species-specific priority-setting using measures of geographic range size. Further details of how I do this are discussed in subsequent data chapters (Chapters 3 and 4), but first it is appropriate to give a brief introduction here as to what is meant by geographic range in the following sub-section.

1.3.1 Geographic range: a central concept for species declines and local rarity

At its simplest, a species' geographic range is the area where a species is found, with the size of this range and how it changes through time making up a fundamental part of a species' ecology and evolutionary history (Gaston, 2003). The size of a species' geographic range has been shown to be a strong predictor of global extinction risk (e.g. Purvis et al., 2000; Cardillo et al., 2006) and is used by the IUCN Red List as a proxy for population declines and a measure of local rarity (IUCN, 2012). Thus, as mentioned in the previous section, species that have small ranges or that have experienced a reduction in the size of their geographic range may be priorities for conservation. Red List criteria that relate to the size of a species' geographic range are the most commonly used and have been used for 47% of the 4440 threatened species of mammals, birds, amphibians and gymnosperms on the Red List (Gaston and Fuller, 2009).

Measuring the size of a species' geographic range is not always easy (Rapoport, 1982; Gaston, 2003). There are two fundamental ways of describing a species geographic range: the Extent of Occurrence (EOO) and the Area of Occupancy (see Figure 1.1). EOO refers to the outermost limit of where the species occurs whereas AOO refers to the actual area that is occupied within the EOO. These two definitions refer to different aspects of geographic range and should not be conflated (Gaston and Fuller, 2009).

In order to construct measurements of geographic range, distribution data is needed. This can come in different forms such as point locality data or the statistical modelling of habitat preferences (Gaston and Fuller, 2009). I use different types of geographic range measure and distribution data in my data chapters and they are described more fully there. As is clear from Figure 1.1, EOO estimates can be particularly affected by data availability at the margins of a species' range (i.e. if data is lacking at the edge of a range, EOO estimates could be underestimated) and AOO estimates can be affected by the choice of spatial resolution (i.e. if extremely coarse spatial resolutions are used in a

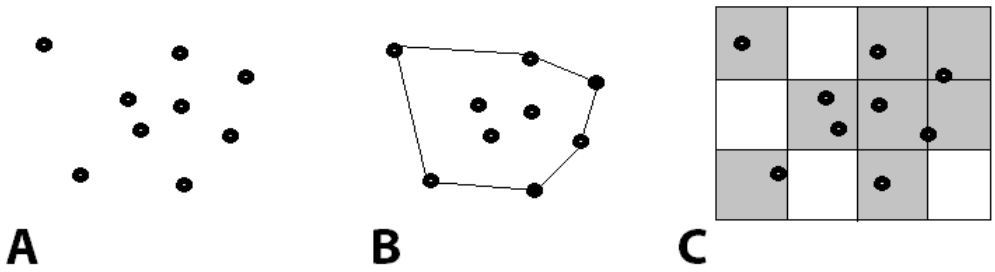


Figure 1.1: An example of the differences between EOO and AOO range measures (after IUCN, 2012). A) shows the spatial distribution of known, inferred or projected sites of present occurrence; B) shows one potential boundary that can be used to construct EOO, which is the measured area within the boundary; C) shows one potential example of using grid cells to construct AOO, which the sum of the occupied (shaded) grid squares.

grid-cell based assessments of AOO, such range size estimates could be overestimated; Gaston and Fuller, 2009).

1.4 Which areas to conserve?

Prioritising conservation resources between species to prevent extinctions can take a variety of approaches. In this thesis I examine approaches that concentrate on the most threatened species or 'fire-fighting' and those that focus more on 'preventive conservation'. Short-term 'fire-fighting' has sometimes been criticised for not considering the cost of conservation or the likelihood of success. Some of these criticisms can be circumvented by targeting action not towards individual threatened species, but towards areas that contain populations of multiple threatened species.

Directing conservation resources towards certain areas in this way is called 'site-based conservation'. This relies on the principle that multiple species may live in the same area and thus may experience the same threats. For example, given that habitat loss threatens over 90% of threatened species (Baillie et al., 2004), giving legal protection to those habitats in which many threatened species live is logical. Typically, this is done through protected areas (PAs), that aim to either halt or manage such threats (Bruner et al., 2001). The IUCN defines a PA as a 'clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values' (Dudley and Stolton, 2008) and recognises up to six different types of PA, each affording different levels of protection (Dudley and Stolton, 2008).

PAs are seen as an important contribution to biodiversity conservation with site-based conservation reported to be appropriate for 82% of birds, mammals

and amphibians (Boyd et al., 2008), although the global PA coverage of species ranges (12.9% of the total land area as of 2009; Jenkins and Joppa, 2009) can be only described as adequate at best (Rodrigues et al., 2004). There has been a significant recent expansion in the number of PAs and the area that they cover in the last 20 years (Jenkins and Joppa, 2009) and the political recognition of the importance of these areas is demonstrated in Aichi target 11 (see Table 1.1), with the intention of extending coverage to 17% of the terrestrial surface (CBD, 2010).

Target 11 makes clear the challenge in assessing how useful protected areas are for biodiversity conservation, as it requires a range of measures to be taken for a meaningful appraisal to be made in addition to percentage targets in global coverage. These measures include ensuring PAs have a suitable extent of representation, especially for areas important for biodiversity and ecosystem services. In addition PAs should provide effective conservation through equitable management and are ecologically representative and well-connected. It is not easy to measure these characteristics so that a useful assessment of the world's 150,000 or so protected areas can be made (Woodley et al., 2012).

In this thesis, I focus on one aspect of Target 11 (see Chapter 5): how well protected areas overlay with biodiversity in a way that reflects the qualities of biodiversity that we wish to conserve. This can be challenging, as different aspects of biodiversity can be valued in different ways (Nunes and van den Bergh, 2001). For example, biodiversity values that are the basis of area prioritisation programmes that have been developed over the last 20 years include endemism (Endemic Bird Areas; Stattersfield, A.J., Crosby, M.J., Long, A.J. and Wege, 1998), overall species richness (hotspots; Myers et al., 2000), representativeness (ecoregions; Olson et al., 2001) and extinction avoidance (Alliance for Zero Extinction sites; www.zeroextinction.org, 2010) to name but a few. Thus, it is clear that many examples of site-based priority setting relate to species-based priority setting and largely fall within the 'vulnerability/irreplaceability' framework introduced earlier in section 1.3 (see Brooks et al., 2006, for a fuller discussion). As a further complication, it has been shown that priority areas as defined by different biodiversity values may be incongruent with each other (Prendergast et al., 1993; van Jaarsveld, 1998; Moritz et al., 2001; Grenyer et al., 2006) and so our assessment of how well PAs overlay with biodiversity can depend on the facet of biodiversity that we are interested in.

Nevertheless, if we are to help curb current and future extinction rates, it is essential to capture biodiversity within the boundaries of protected areas (Pimm and Raven, 2000; Schipper et al., 2008). Ensuring that the desired quality of biodiversity we wish to conserve is represented in PAs in an optimal

way is called 'systematic conservation planning' (Margules and Pressey, 2000). I explore the issues of representing different qualities of biodiversity through systematic conservation planning further in Chapter 5.

1.5 Data and decision-making

There are a variety of prioritisation approaches available that helps to conserve biodiversity but I have introduced those that examine species and those that examine sites, although in practise a mixture of both are often used. However, for both approaches and conservation in general, there is often insufficient data on both the extinction threat species face and where species are distributed to be able to make meaningful and informed priority-setting decisions (Kozlowski, 2008). This data shortfall is not trivial because it could negatively impact both our ability to achieve and measure our progress towards our Aichi biodiversity commitments (Kozlowski, 2008).

As I have outlined earlier (see section 1.3), species distributions (i.e. geographic range) form a core part of the IUCN's Red List process for assessing a species' extinction risk. I briefly explain why a lack of data on where species are found and their global extinction risk can impact on conservation-priority and outline a potential solution in the following sub-sections.

1.5.1 Data shortfall: species distributions

Our lack of knowledge of where species occur is sometimes termed the 'Wallacean shortfall' (Lomolino, 2004). This occurs across all spatial scales (Lomolino, 2004; Whittaker et al., 2005) and is thought to be largely due to insufficient financial investment in field surveys and data inventories (Kozlowski, 2008). One major consequence is that uncertainties in the spatial distribution of biodiversity may not be acknowledged in systematic spatial conservation planning (Polasky et al., 2000; Gaston and Rodrigues, 2003; Rocchini et al., 2011), which may lead to PA networks being placed in the wrong places and thus reducing the efficacy of PAs in halting biodiversity loss (Rondinini et al., 2006).

1.5.2 Data shortfall: species extinction risk

In addition to affecting systematic conservation planning, the Wallacean shortfall has been implicated in the lack of data on species extinction risk along with seven other factors, such shortfalls in our knowledge of the threats species face (Kozlowski, 2008) and more fundamentally, a lack of knowledge of the existence of some species themselves (sometimes termed the 'Linnean

shortfall'; Lomolino, 2004; Whittaker et al., 2005) – a fact implicitly acknowledged in Aichi target 12. There is also often a lack of data on the abundance of species and how it changes in space or time, which is sometimes called the 'Prestonian shortfall' (Cardoso et al., 2011). Thus, given that species' distributions (i.e. geographic range) and measures of population abundance are used in the IUCN process for assessing extinction risk, inaccurate or incomplete distributions data and abundance data can affect assessments of species declines, local rarity and thus our assessment of relative extinction risk between species.

An incorrect assessment of extinction risk can have large ramifications for how we prioritise species-specific funding. For example, this can have large effects on conservation triage (see section 1.3) and affect when we decide to give up investing in rare species.

1.5.3 Historical data: part of the solution?

Thus one of the major problems conservation biology faces is a lack of data on species' extinction risk and species distributions. Given that conservation is underfunded, this problem is unlikely to go away. So, how are conservationists to act in the face of uncertainty? One approach has been to collect more data, but this is often not the best course of action (Legg and Nagy, 2006; McDonald-Madden et al., 2010) and it has been argued that hesitation in the face of incomplete data delays management interventions (Lindenmayer et al., 2013) and can fail to prevent biodiversity loss (Martin et al., 2012). Recently one paper has advocated we adopt an approach from mathematics; thus, rather than trying to solve these problems through increased data collection, we should aim to work 'backwards' to see what variations on the components might achieve the desired outcome (Grose, 2014).

Another potential approach is to use previously disregarded data sources, such as historical occurrence data. Tingley and Beissinger (2009) define historical occurrence data as: 'any set of information that through observed detections or non-detections, provides evidence on the true presence or absence of individuals of a species' and may include museum specimens, sighting records and atlas data. Using such data may be challenging (Tingley and Beissinger, 2009) due to acknowledged potential pitfalls (summarised in Table 1.2), which are the result of the opportunistic and non-systematic way that historical data is collected (e.g. Boakes et al., 2010). However, if these pitfalls are acknowledged and accounted for, historical data can be usefully applied to a range of conservation problems: for example, it can be used to help provide greater information on species declines (i.e. to help identify long-term declines;

Shaffer et al., 1998), species extinction status (i.e. through extinction probability calculations; Collen et al., 2010) and species distributions (i.e. through species distribution modelling; Raxworthy et al., 2003). It has also been used to investigate conservation measures, such as the efficacy of PAs in preventing habitat loss (Clark et al., 2013).

Given the rapid pace of environmental change and the potential impact of data shortages on conservation priority-setting, it is arguable that all potential sources of information need to be examined. In this thesis I make use of a historical dataset, which I describe in more detail later.

Problem	Description
Error	Non-specimen records may have been identified incorrectly depending on observer skill; associated locality information may be incorrect
Biases	Different areas and time periods have more historical record collecting than others and different attitudes to collecting
Presence vs. Absence data	Non-systematic and opportunistic collection of data means it is often impossible to distinguish true absences from a lack of survey effort or failure to account for detectability; historical presences may indicate demographic sinks and thus low quality rather than favoured habitat

Table 1.2: Some of the main problems in using historical occurrence data (Adapted from Graham et al., 2004).

1.6 Greater Himalayan biodiversity in crisis

Global biodiversity is in a state of crisis and humanity has committed itself through international law to an improvement in the status of biodiversity. Given limited funding, conservation resources need to be directed towards priority species and priority areas if we are to achieve these targets. However, this is sometimes difficult to implement due to a lack of data. Having introduced these broad concepts, I now focus on how they apply to one priority area of conservation concern and one highly threatened taxonomic group: species from the bird order Galliformes that live within the Greater Himalaya.

The Himalayan mountain range was created by the convergence of the Indian and Eurasian continental plate margins approximately 65-70 million years ago (Xu et al., 2009), profoundly affecting the geology of a huge swathe of South East Asia. Today, the Greater Himalayan region covers around seven million square kilometres of high mountains and plateaus within Central, Southern and Inner Asia (Qiu, 2008; Xu et al., 2009), spanning around seven countries (for a summary of the Himalayan countries see Table A.1 and

Figure A.1). Details on how I delimited the precise boundary of the Greater Himalayan region are in the thesis Appendix.

The underlying geology of the region has had profound effects on its biodiversity, with the level of species richness in the Greater Himalaya far exceeding that of the surrounding lowlands (Xu et al., 2009). The causal mechanism that has driven this pattern is thought have been the rising mountains, which have provided physical barriers to gene flow and promoted speciation via climatic-zone compression along altitudinal gradients, exposure effects and high levels of micro-habitat diversity (Xu et al., 2009). Simultaneously, the many peaks and valleys have also acted as a refuge, affording any flora or fauna respite from extinction events that have occurred in the lowlands. Thus, the geology of the region has helped to both maintain and generate an exceptional number of geographically restricted or endemic species. This dual source of endemism is thought to explain why the absolute number of species (or species richness) in the Greater Himalaya is higher than in the lowlands, which is in contrast to the general trend seen in many other alpine environments where species richness tends to decrease with increased elevation (Korner, 2004; Salick et al., 2004).

The exceptional levels of endemism contained within the Greater Himalaya have been formally recognised, with three of the world's 35 global biodiversity hotspots spatially intersecting the region (Mittermeier et al., 2004). However, in order to qualify as a biodiversity hotspot, a region must not only contain at least 1500 endemic species of vascular plants, but must also have lost 70% of its primary vegetation (Myers et al., 2000). Thus, much of the unique biodiversity contained within the Greater Himalaya is currently under threat. This is further exemplified by the fact that the region has been identified as having a disproportionately high number of threatened terrestrial species, with a net increase in the overall extinction risk of birds, mammals and amphibians (Hoffmann et al., 2010).

The poor current state of biodiversity within the Greater Himalaya is almost certainly due to anthropogenic influences. While the Greater Himalaya has historically remained relatively isolated from most human activity, this is no longer the case and many Himalayan countries have experienced rapid population growth (Bawa and Dayanandan, 1997), which has resulted in the concomitant rise in the number of anthropogenic threats to biodiversity. The nature of these anthropogenic threats is varied and ranges from overhunting to pollution. However, the conversion of natural habitats is thought to be by far the largest and most important driver of Himalayan biodiversity loss.

Unfortunately, these anthropogenic pressures are unlikely to abate in the

future, with one quarter of endemic species (birds, mammals and amphibians) predicted to be lost due to deforestation by the year 2100 in the Indian Himalaya alone (Pandit et al., 2006). Similarly, rising population projections from the UN for Himalayan countries suggest that humans are likely to continue to place more pressure on the wildlife contained there (United Nations Department of Economic and Social Affairs Population Division, 2013). While some existing threats are likely to continue to shape future biodiversity patterns, other novel threats are also likely to make their impact increasingly felt and may act in synergy with each other (Sala et al., 2000; Brook et al., 2008), such as anthropogenic climate change. Climate change has been predicted to be of importance for high altitude species (La Sorte and Jetz, 2010). The effects are likely to range from a narrowing in suitable habitat (which is known as being trapped on 'sky islands' (Heald, 1967)), to increased nitrogen deposition and an increased frequency of forest fires (Beniston, 2003). Many of these effects are likely to be far reaching and to have major ramifications not only for Himalayan biodiversity (which has been termed 'an impending disaster'; Sodhi et al., 2004), but also for humans that rely on the ecosystems Himalayan biodiversity provides, such as helping to prevent floods (Xu et al., 2009).

1.7 Himalayan Galliformes

One key taxonomic group that is found throughout the Greater Himalaya in its entirety is the bird order Galliformes (gamebirds), which includes species of pheasant, partridge and quail. In line with regional biodiversity patterns for other taxa, the Greater Himalaya is particularly species-rich in Galliformes containing 24 resident species, of which 18 are endemic to the region and found nowhere else on earth. In this thesis I focus on this set of resident species that are found in the study region throughout the year (ENVIS, 2007; BirdLife International and NatureServe, 2011) and not on vagrants or introduced species (see Figure A.2 in the thesis Appendix for images of each species and Table A.2 in the thesis Appendix for a full list of each species).

Galliformes are an important group of birds for humans that live within the Himalaya both culturally and from a nutritional perspective. Galliform species are listed as national birds for three Himalayan countries: the Himalayan monal (*Lophophorus impejanus*) is the national bird of Nepal, the common peafowl (*Pavo cristatus*) is the national bird of India and chukar partridge (*Alectoris chukar*) is the national bird of Pakistan. The ornate plumage of many Galliformes species has meant that such species often play an important role in the folklore and cultures of Himalayan people, with the common peafowl a noticeable example (Fitzpatrick, 1923; Thankappa, 1974). In a more practical

sense, Galliformes are also important as a source of protein for humans within the Himalaya, either through the rearing and husbandry of domestic chickens (*Gallus gallus*), or the hunting of its wild relatives (Kaul et al., 2004; Keane et al., 2005).

The Galliformes that live in the Himalaya are a diverse suite of species: they vary greatly in ecological characteristics, geographic distribution patterns, abundance levels and extinction risk (see Figure A.3 and Table A.2 in the thesis Appendix for further details). For example, some species are widely distributed generalists found across a wide range of habitats across the globe (such as the chukar partridge) and some species are narrowly restricted specialists restricted to alpine habitats in the Himalaya (such as the Himalayan monal). Similarly, some species may be locally abundant but have a high risk of global extinction (such as cheer pheasant; *Catreus wallechi*) and other species may be locally scarce and have a low risk of global extinction (such as common quail; *Coturnix coturnix*). One species, the Himalayan quail (*Ophrysia superciliosa*), is both locally scarce and has a high risk of extinction. The Galliformes order is the most threatened of all bird orders (Bennett and Owens, 1997; Keane et al., 2005) and those species found in the Himalaya face multiple threats to their continued survival (see Figure A.3 of the thesis Appendix). Many of these threats are non-specific threats that affect multiple taxonomic groups, such as habitat conversion (Sodhi et al., 2004), but others may be targeted towards Galliformes in particular, such as hunting (Kaul et al., 2004).

1.8 The central challenge: how to conserve Himalayan Galliformes

Despite the importance of Himalayan Galliformes both from an ecological and human standpoint, our current knowledge of this threatened taxonomic group contains many gaps. For example, it is suspected that many Himalayan Galliformes may have undergone marked population declines, but as to why they have declined, to what extent they have declined and even the current extinction status of some species is still not well understood. There is similar paucity of knowledge regarding how best to target conservation action in the Himalaya for Galliformes. For example, action may need to be targeted across different spatial scales, multiple countries and towards specific areas and species. It may be that using the Red List to set species-specific priorities is not the best course of action. Compounding this further is a poor knowledge of where Galliformes live in the Himalaya; the remote, high peaks make field surveys difficult and it can be hard to disentangle species absences from a lack of survey effort. A newly available database of historical records collected by researchers at Imperial College London (Boakes et al., 2010) has the potential to

help tackle some of these conservation problems. Given the problems of a data shortfall in conservation biology as outlined earlier (see section 1.5), discovering the utility of this database is extremely valuable given the known difficulties in using historical data (see Table 1.2).

In addition to being an important topic for study in its own right, investigating the declines and the conservation of Himalayan Galliformes is valuable because it acts as a microcosm for broader conservation issues such as: which species and areas should we prioritise and how can we achieve this using limited conservation data? Examining broad conservation issues at smaller spatial scales in a bottom-up fashion and in a local context has been recognised as being essential to the implementation of conservation action at the global scale and can thus help us to achieve global biodiversity targets (Mace et al., 2000; Whittaker et al., 2005; Brooks et al., 2006).

1.9 Aims of thesis

The purpose of this thesis is to help fill some of these research gaps with the ultimate aim of assessing how best to direct resources for Galliformes in the Himalaya using a new database of historical records. Given the variety of conservation prioritisation approaches outlined earlier, I focus on just a few examples. Specific concepts and methods are explained in greater detail in each individual data chapter.

The individual aims of my thesis are to:

1. map current distribution of Himalayan Galliformes
2. map past geographic ranges and compare them with more contemporary distributions to explore patterns of long-term range change
3. compare global range changes of narrowly distributed and widespread species
4. identify whether current spatial conservation measures are optimal for Himalayan Galliformes

1.10 Thesis outline

In order to achieve these aims, I first investigated and described the data contained within the new database (see thesis Appendix for a full description).

Subsequently, I start the thesis with just one species of Galliform in the Himalaya, the Himalayan quail. Here, I use scarce data to predict whether this species is likely to be extinct and where field searches stand the best chances of

finding it by creating a new map of its potential distribution, thus fulfilling Aim One. This is an example of how we can direct conservation efforts for a highly threatened species, even though this may not be the best use of conservation resources in terms of minimising species extinctions overall. The Himalayan quail is the most threatened Galliform in the Himalaya, if not the most threatened species and it presents a unique range of analytical challenges that require it to be treated on its own.

For the rest of my thesis, I examine aspects of conservation priority-setting across multiple species and sites. First, I look at species distributions for all Himalayan Galliformes but in a different way to the previous chapter. Specifically, I examine differences between past and present distributions, or range changes. I use the results to ask questions, such as: are patterns of range change inside the Himalaya greater than outside the Himalaya and if so, how do those patterns relate to extinction risk as indicated by the Red List? This fulfils both Aims One and Two and is an example of how it can be useful to examine both conservation 'fire-fighting' and 'preventive conservation' approaches when setting species-specific priorities.

The results from the aforementioned analysis raises an interesting biogeographic point: are endemic species or species with small geographic ranges more or less likely to have undergone range declines? I examine the difference between past and present distributions as before but for all Galliformes, rather than a Himalayan subset. By examining an entire taxonomic group in this way, I am able to make a broad assessment of a biogeographic pattern that could help inform the way we assess the conservation status of widespread species across the globe today. At a local level, I comment on whether specific changes need to be made to the way we assess both declines and conservation status in widespread Himalayan Galliformes. This fulfils Aims Three and is a further investigation into setting species-specific conservation priorities.

Next, I return to Himalayan Galliformes and map their current species distributions, overlaying them with the current Himalayan protected area network to assess how well such distributions are protected spatially. Different ways of prioritising conservation efforts between species and sites while accounting for data uncertainty are examined. This fulfils both Aims One and Four and is an example of site-based priority-setting.

Finally I discuss the results from my thesis in both a local and global context, including practical conservation implications for both Himalayan Galliformes and the Aichi biodiversity targets, as well as identifying future research directions and making a series of recommendations.

Chapter 2. Mapping the potential distribution of the Critically Endangered Himalayan Quail (*Ophrysia superciliosa*) using proxy species and species distribution modelling

2.1 Abstract

The Critically Endangered Himalayan quail (*Ophrysia superciliosa*) has not been reliably recorded since 1876. Recent searches of historical sites have failed to detect the species, but we estimate an extinction year of 2023 giving us reason to believe that the species may still be extant. Species distribution models can act as a guide for survey efforts, but the current land cover in the historical specimen record locations is unlikely to reflect Himalayan quail habitat preferences due to extensive modifications. Thus, we investigate the use of two proxy species: cheer pheasant (*Catreus wallechi*) and Himalayan monal (*Lophophorus impejanus*) that taken together are thought to have macro-habitat requirements that encapsulate those of the Himalayan quail. After modelling climate and topography space for the Himalayan quail and these proxy species, we find that the models for the proxy species have moderate overlap with that of the Himalayan quail. Models improved with the incorporation of land cover data and when these were overlaid with the Himalayan quail climate model, we were able to identify suitable areas to target surveys. Using a measure of search effort from recent observations of other Galliformes, we identify 923 km^2 of suitable habitat surrounding Mussoorie in Northern India that requires further surveys. We conclude with a list of five priority survey sites as a starting point.

2.2 Introduction

Species that are faced with an imminent risk of extinction are conservation priorities (Vane-Wright, 2009; IUCN, 2012). Such species are typically rare and low in numbers (Gaston, 1994), which makes them difficult to detect and as a result, difficult to assess their status with confidence. Consequently, assessments are often reliant on subjective expert opinion (van der Ree and McCarthy, 2005). The monophyletic Himalayan quail (*Ophyrisa superciliosa*) is Critically Endangered (IUCN, 2012) due to a small putative population, a narrow geographic distribution and intensive habitat modifications (IUCN,

2012). Quantitative assessment of the species' status and potential distribution is essential for the targeting of efforts to rediscover the species and hence its conservation.

Few records of the Himalayan quail exist: the last specimens date from 137 years ago and there has been a lack of confirmed records since. Re-sighting probability calculations offer a more objective, probabilistic insight into extinction assessments as we have little evidence for, or confidence in declarations of extinction based on the raw data alone (Butchart et al., 2006). A recent estimate for Himalayan quail suggests it went extinct in the late 1890s (Collen et al., 2010), only c.20 years after the last of a small number of sightings. Recently, further reports have been collated and all records for the species have been subjected to a critical re-evaluation (Boakes et al., 2010). Consequently, a revised estimate of extinction is needed. If the species is extant, search effort needs to be targeted to the most suitable areas.

Specimen records for the Himalayan quail come from Mussoorie and Nainital, in the state of Uttarachand, Northern India. Surveys undertaken to date have failed to detect the Himalayan quail around these areas and further surveys in western areas of Nepal have also not located the species (Cuthbert, unpublished data.). The lack of modern records suggests the need for enhanced techniques in these areas and/or searches in additional areas where the species may still occur. However, the distribution of the species is uncertain and its geographic range is all but unknown or inaccurately mapped (e.g. BirdLife International, 2014b). An updated and a quantitative assessment of its range made using the best available data is needed if surveys are to be appropriately targeted.

For most rare species, surveys are undertaken in areas of most suitable habitat to maximise the likelihood of detection. One approach to identifying such habitat is to generate species distribution models (e.g. Guisan and Zimmermann, 2000; Boitani et al., 2011). However, while historical records may still reflect the climate space of the species, they are unlikely to reflect its habitat preferences because reliable records come from historical areas that have been extensively modified, creating a temporal mismatch between specimen records and habitat covariates. Other approaches are therefore needed for this species. The use of proxy species is well-documented in conservation biology (Caro et al., 2005) but the use of proxy species to identify suitable habitat is novel. Here we use more abundant and widespread species that might share similar habitat requirements as a guide for directing survey efforts. Cheer pheasant (*Catreus wallechi*) and Himalayan monal (*Lophophorus impejanus*) have distributions that overlap the known range of the Himalayan quail and utilise areas of dense grass that are potentially similar to the habitat requirements of the Himalayan quail (Kaul, 1992; Del Hoyo, J., Elliott, A. and

Sargatal, 2001; BirdLife International, 2014a). Although extensive knowledge of the Himalayan quail's habitat preferences is absent, the proxy species used to identify suitable habitat must be well-justified, especially if they occupy a much wider geographic distribution than that of the target species.

We attempt to update our knowledge of the potential distribution of the Himalayan quail by generating a climate and topography model for the Himalayan quail, the Himalayan monal and cheer pheasant using Maximum Entropy (Maxent) species distribution modelling software (Phillips et al., 2006; Phillips and Dudík, 2008). Generating a climate and topography model for the Himalayan quail is useful because these parameters are less likely to have changed than land cover and therefore likely to represent a realistic niche for this species. We next test the overlap of the Himalayan monal and cheer pheasant climate models with that of the Himalayan quail to assess whether these two species can be used as potential proxies. Overlap between the climate and topography model of the Himalayan quail and that of the proxy species would indicate that they occupy similar climate spaces and thus reflects the suitability of the proxy species as surrogates. Subsequently, we generate full species distribution models including habitat for the proxy species, which are reliable because there are modern records for these species. Overlap between the full habitat models for the proxy species and the climate model for Himalayan quail will indicate potentially suitable areas for the species. We refine this area of suitable habitat further by incorporating measures of previous survey efforts to identify areas that have been poorly searched.

2.3 Methods

2.3.1 Bird records

We took data from the GALLIFORM: Eurasian Database V.10 (Boakes et al., 2010) that contained point locality data accurate from 0.62 - 30 miles (1 - 48.3 km; for further details see Tables 2.A1 - 2.A3 in chapter appendix section 2.6.1), collected from a wide-range of sources including museum specimens, ringing records, biological atlas data and trip reports (see Figure 2.1 panel A and Table 2.A1 in chapter appendix section 2.6.1).

We omitted records if they lacked latitude and longitude coordinates and a date (19 records were omitted in this way). Given the uncertainty around the reliability of the contemporary records, we also undertook a preliminary vetting procedure to ensure that the Himalayan quail data used in the creation of our species distribution models was suitable. Thus, we omitted two records collected post-1980 that were 20 km from Mussoorie and Nainital and lacked

information on record type (i.e. whether they were a primary or secondary record) and observation type (i.e. whether they were sightings or indirect signs etc.) as it was impossible to judge the reliability of these two records (for further details of the number and nature of records considered, see Table 2.A4 in chapter appendix section 2.6.1). All other records were used along with the historical specimen records in the climate model. This reduced our sample size for Himalayan quail from 55 to 34 records.

We used all available post-1980 records in the cheer pheasant and Himalayan monal models, as these species are well-known and readily-identified. The cheer pheasant has already been identified as a potential proxy for the Himalayan quail (Kaul, 1992) but the use of the monal in this way is new and is based upon field insights of its habitat use (Cuthbert, unpublished data). In contrast to the cheer pheasant, the monal is known to be an altitudinal migrant (BirdLife International, 2014a), leading us to believe that the summer distribution of the monal might be most similar to that of the Himalayan quail. Accordingly we first attempted to model the summer distribution of the monal but also modelled the annual distribution of the monal in case the resulting summer climate model overlapped poorly with that of the Himalayan quail. Summer was defined two ways: 1) on the basis of elevation i.e. records below 3800 m (the mid-point of the Himalayan monal's described altitudinal range) were classified as winter records (N = 249) and those above 3800 m (N = 68) were classified as summer records (Ali and Ripley, 1983); 2) on the basis of specimen labels i.e. records were labelled summer (N = 12) or winter (N = 54) records. We used both approaches because although the former was more subjective, not all records were labelled according to the season when they were collected and we wanted to include as much point locality data in our models as possible to ensure they gave both accurate and precise predictions.

2.3.2 Calculating the likelihood of extinction

We used Optimal Linear Estimation (OLE; Cooke, 1980) to assess the probability that the Himalayan quail was globally extinct. OLE is a technique that is commonly used to assess the likelihood of extirpation (Roberts and Solow, 2003; Solow, 2005). OLE is a non-parametric method that allows extinction dates to be estimated based on the distribution of the most recent sightings. The main assumption is that the sighting effort never falls to zero over an annual time step, particularly around the time of extinction. OLE is known to be sensitive to the number of records included in the calculation, but for species with more than five sightings, it is thought to provide robust estimates for the time of extinction (Collen et al., 2010). To ensure our estimates were robust, we included all records, records from the last five years before

(and including) the final sighting and specimen records only (see Table 2.A5 in chapter appendix section 2.6.2). Calculations were undertaken in R (R Development Core Team, 2012) using the R package 'sExtinct' (Clements, 2013).

2.3.3 Modelling approach for climate envelopes and species distribution models

We created two sets of species distribution models: climate and topography models for the Himalayan quail and proxy species and then full models also including land cover for the proxy species only. We used MaxEnt (version 3.3.3k; Phillips et al., 2006) to model the likelihood of occurrence of the species using presence locations of each species in turn as a function of topography and climate for the first set of models and topography, climate and land cover for the second set of models. Maxent has been found to perform well against other distribution models (Elith and Graham, 2009) and produces models that have particularly high accuracy in the case of species with small sample sizes and restricted geographic locations. Rather than use MaxEnt's default setting of selecting 10,000 randomly generated pseudo-absences, we used locations from which there were records from other Galliformes (N = 820 records) to generate a targeted set of pseudo absences. Thus, our pseudo-absences were chosen from sites with the same sampling bias as the presence sites for a suite of species that were observed with similar sampling techniques. This "target group" approach (Phillips and Dudík, 2008) reduces the potential for species distribution model outputs to be affected by sampling biases in study species records in both time and space (see Boakes et al., 2010, for a description of the sampling biases in the dataset that we use).

For each species, we omitted duplicate records to identify a subset suitable for inclusion in our final species distribution models. Analysis was restricted to occupied and immediately surrounding ecoregions for each modelled species. By restricting the analysis in this way, we ensure that we obtain outputs that are sensible in their geographic distribution (for further details of the Himalayan ecoregions used in our analysis and the distribution of records see Table 2.A6 in chapter appendix section 2.6.3).

Climate variables were downloaded from www.worldclim.org/bioclim (2014) and were approximately 1 km^2 in resolution. In an effort to reduce the number of variables considered (and thus reduce the risk of model over-fitting), we only considered four of the 17 potential climate variables relating to temperature and precipitation that described the major axes that are likely to affect the distribution of our three bird species. These were annual mean temperature, temperature seasonality, average precipitation and precipitation seasonality. Elevation was downloaded from the 90 m Shuttle Radar

Topography Mission (SRTM) at 30 arc seconds (Jarvis et al., 2008) and was used to assess altitude, slope and aspect. Slope and aspect rasters were created and standardised in ArcMap version 10.2.

For the full proxy species distribution models we also incorporated a measure of Normalised Difference Vegetation Index (NDVI), a continuous measure of habitat type. NDVI data collected by the SPOT-Vegetation platform from 1999-2007 were downloaded from www.vito.be and the middle dekad (10-day period) of each month was extracted and averaged across years using raster calculator in ArcGIS version 10.2 and clipped to the modelled area using VGT-Extract. The rasters were standardised to the same resolution (1 km^2) as before. No attempts were made to omit collinear variables as machine learning methods have been shown to still perform well with such variables, especially when the study goal is predictive accuracy (Elith et al., 2011).

Variables selected for inclusion in the final proxy models were those that contributed >3% to the maximal model to avoid over-fitting the models while maximising their predictive power (this threshold was chosen after trial and error; for further model details see Table 2.A7 in chapter appendix section 2.6.3). Optimal feature functions were chosen based on sample size (Phillips and Dudík, 2008) and regularisation parameters were chosen based on AICc (Warren and Seifert, 2011). All final models (see Table 2.A8) were clipped to the occupied and neighbouring ecoregions of Himalayan quail to maintain a focus on the Himalayan quail climate map.

The ability of each model to discriminate between occupied and unoccupied areas was estimated from the area under the curve (AUC) of the receiver operating characteristics (ROC; Phillips et al., 2006). We used cross-validations to generate ten folds of randomly-selected presence data and ran the model ten times, excluding each fold in turn and using the fold to validate the model (Phillips and Dudík, 2008). This uses all the data for validation and allows the standard deviation of the mean AUC to be assessed.

A minority of the point locality data were collected at a larger spatial resolution than our predictor variables (see Tables 2.A1 - 2.A3 in chapter appendix section 2.6.1) with the capacity to bias our outputs. However, the use of data with 1 km^2 accuracy did not change the high AUC value for our SDMs (the difference in mean AUC values for models based on all data vs. models based on accurate data only were +0.01 for the full model of the cheer pheasant, 0.00 for the full model of the Himalayan monal and 0.00 for the climate model of the Himalayan quail).

2.3.4 Comparing climate envelopes

The continuous model outputs for each species were categorised as suitable or unsuitable based on a threshold derived from the equal training sensitivity and specificity score. This threshold was chosen because it has been shown to minimise the rate of false positives and negatives (Pearson et al., 2004).

We compared climate space between the Himalayan quail and proxy species by identifying thresholded climate space for Himalayan quail from the western part of the range and adding a minimum convex polygon to create a comparison area of approximately 20,000 km^2. This was done to ensure a focus on the Himalayan quail's climate space and to ensure a fair comparison was made. We then clipped our Himalayan monal and cheer pheasant climate outputs to this area and calculated a spearman rank correlation for the continuous logistic output using ENMTools and the Kappa statistic, a measure of spatial agreement for the categorical thresholded output using Map Comparison Kit 3.2.3 (Visser and de Nus, 2006). Finally, we assessed whether the Himalayan quail was restricted in its climatic envelope by examining the shape of the modelled response curves.

2.3.5 Identifying overlap between climate and niche envelopes

The climate map of the Himalayan quail and the full ENM maps for the proxy species were combined using raster calculator in ArcGIS version 10.2. The maps were projected using a South Asia Albers equal area projection. Areas of suitable climate for the Himalayan quail that overlapped with suitable habitat for the respective proxy species indicated potential areas for surveys.

Additionally, areas that overlapped with the proxy species were further refined by a measure of search effort. This was undertaken by creating a kernel density raster based on the number of post-1980 records from all Galliformes species in the area per square kilometre and divided into five geometric intervals. The localities previously identified by the overlap analysis were then multiplied by this raster to further refine priority sites. Thus, of the suitable areas based on climate and habitat suitability, those with the lowest search effort may be those in highest need of surveying.

2.4 Results

When all reliable records were used, our Optimal Linear Estimation calculation estimated the time of extinction to be the year 2023 (CI 1999-2120; Figure 2.1 panel B and Table 2.A5 in chapter appendix 2.6.2). Based on the four

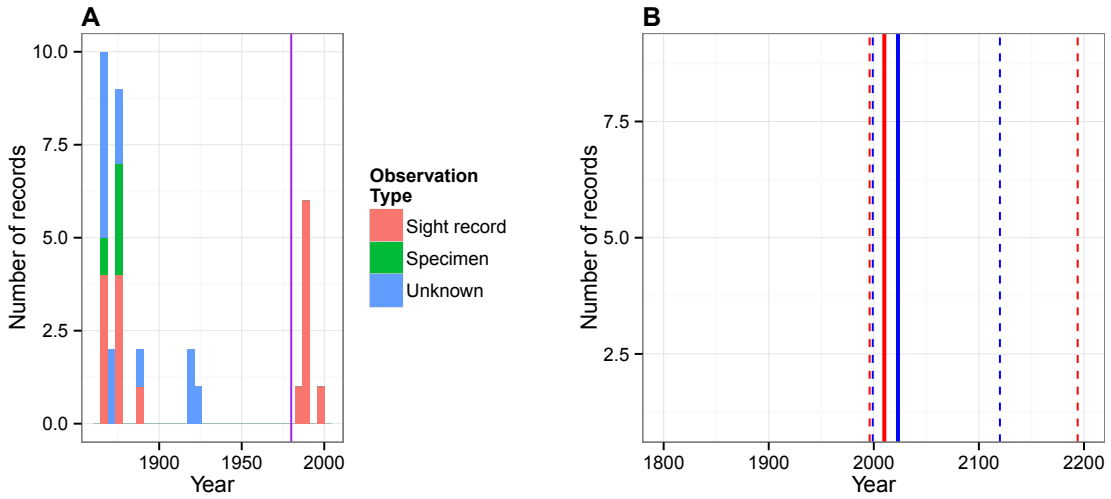


Figure 2.1: Plots of Himalayan quail records through time and extinction date estimates. A) is a histogram showing the number of records across time and the type of observation record. The purple line represents the year 1980. There are no specimen records post-1880. B) shows Optimal Linear Estimation extinction dates. Blue = all records used in the calculation. Red = records from the last five years before (and including) the final sighting used in calculation. Dashed lines indicate upper and lower 95% confidence intervals respectively.

bioclimatic variables used, Himalayan quail had the narrowest climate distribution, followed by cheer pheasant and Himalayan monal (see Figure 2.2 panel A). Variation in temperature and precipitation best predicted cheer pheasant potential occupancy (see Table 2.A9 in chapter appendix section 2.6.3); for Himalayan monal it was highest based on variation in temperature, elevation, mean annual temperature, mean annual precipitation, temperature variation and slope (see Table 2.A10 in chapter appendix section 2.6.3); for Himalayan quail it was highest based on variation in temperature, mean annual temperature and elevation (see Table 2.A11 in chapter appendix section 2.6.3). The Himalayan monal climate envelope created from all records was more tightly associated with the Himalayan quail's climate enveloped than the Himalayan monal climate envelopes based on summer records (see Table 2.1), so all records were used for the Himalayan monal's final niche envelope.

Comparison species	Continuous			Categorical			
	Spearman r^2	Spearman slope	Spearman co-ef	Kappa	K_{loc}	K_{hist}	Fraction agreement
Cheer pheasant*	0.66	0.58	0.32	0.12	0.91	0.14	0.43
Himalayan monal (all)*	0.72	0.99	0.25	0.43	0.83	0.52	0.71
Himalayan monal (summer labelled)	0.23	0.53	0.61	0.00	-0.40	0.00	0.71
Himalayan monal (summer partitioned)	0.22	0.43	0.61	0.06	0.87	0.07	0.73

Table 2.1: Climate overlap between Himalayan quail and proxy species as assessed using a) continuous output (Spearman rank) and b) categorical output (Kappa). Values of kappa <0.2 indicate slight agreement, whereas those between 0.4-0.6 indicate moderate agreement (Landis and Koch, 1977). Asterisks (*) indicate the species and record subset used in the full niche models.

The likelihood of occupancy from the cheer pheasant and Himalayan monal climate model was positively correlated to that of the Himalayan quail's climate model (see Table 2.1) suggesting the likelihood of occupancy in shared locations increased in a similar way for both species. The Kappa statistics revealed slight (<0.2) inter-rater agreement between the Himalayan quail and cheer pheasant and moderate (0.4-0.6) inter-rater agreement between the Himalayan quail and Himalayan monal model. Taken together, Himalayan monal appears to be a better proxy for Himalayan quail by way of climate than cheer pheasant.

2.4.1 Creating niche envelopes

We produced niche envelopes (see Figures 2.2 panel B and 2.2 panel C) for the two proxy species that incorporated additional land cover variables. They represented suitable areas of 104,228 km^2 for cheer pheasant and 162,249 km^2 for Himalayan monal. In comparison to the climate/topography models, this corresponded to an increase in suitable area of 1.2% and 150% for cheer pheasant and Himalayan monal respectively. The mean AUC for cheer pheasant was 0.89 (s.d. = 0.02) and for Himalayan monal was 0.80 (s.d = 0.06). Predicted likelihood of occupancy for cheer pheasant was highest with variation in temperature, June, January, December NDVI and elevation. Predicted likelihood of occupancy for Himalayan monal was highest with variation in precipitation, February, June, July, August NDVI, elevation, mean annual temperature, variation in temperature and slope. For full details of the relative importance of model covariates see Tables 2.A12 and 2.A13. We produced the following combined map (see Figure 2.3 panel A) to show locations where the proxy species overlapped with the Himalayan quail (an area of 8607 km^2). Of the total Himalayan quail cells with a modelled value exceeding the logistic threshold of equal training sensitivity and specificity (9734 km^2), 43.5% overlapped with both proxy species, 1.6% with Himalayan monal only, 43.3% with cheer pheasant only and 11.6% had no overlap with either of the proxy species. Of the localities that overlapped with both proxy species, 924 km^2 (23%) were in areas with low levels of survey effort (6-40 records/km^2), particularly in the North West India near to but outside of Mussoorie, Uttarachand (see Figure 2.3 panel B and Table 2.2).

Location name	Latitude	Longitude
Bhimleth	30°32′13″N	78°33′48″E
Khasonsi	30°37′22″N	78°08′47″E
Tyongi Pangu	29°59′41″N	80°38′43″E
Dug R.F.	29°58′02″N	80°29′18″E
Chirbitiyakhal	30°23′12″N	78°50′05″E

Table 2.2: The top five priority localities for surveys of the Himalayan quail. All localities are in Uttarakhand, India and latitude/longitude co-ordinates refer to villages. Searches should be made within surrounding grassy scrubland areas on steep slopes outside of the villages themselves.

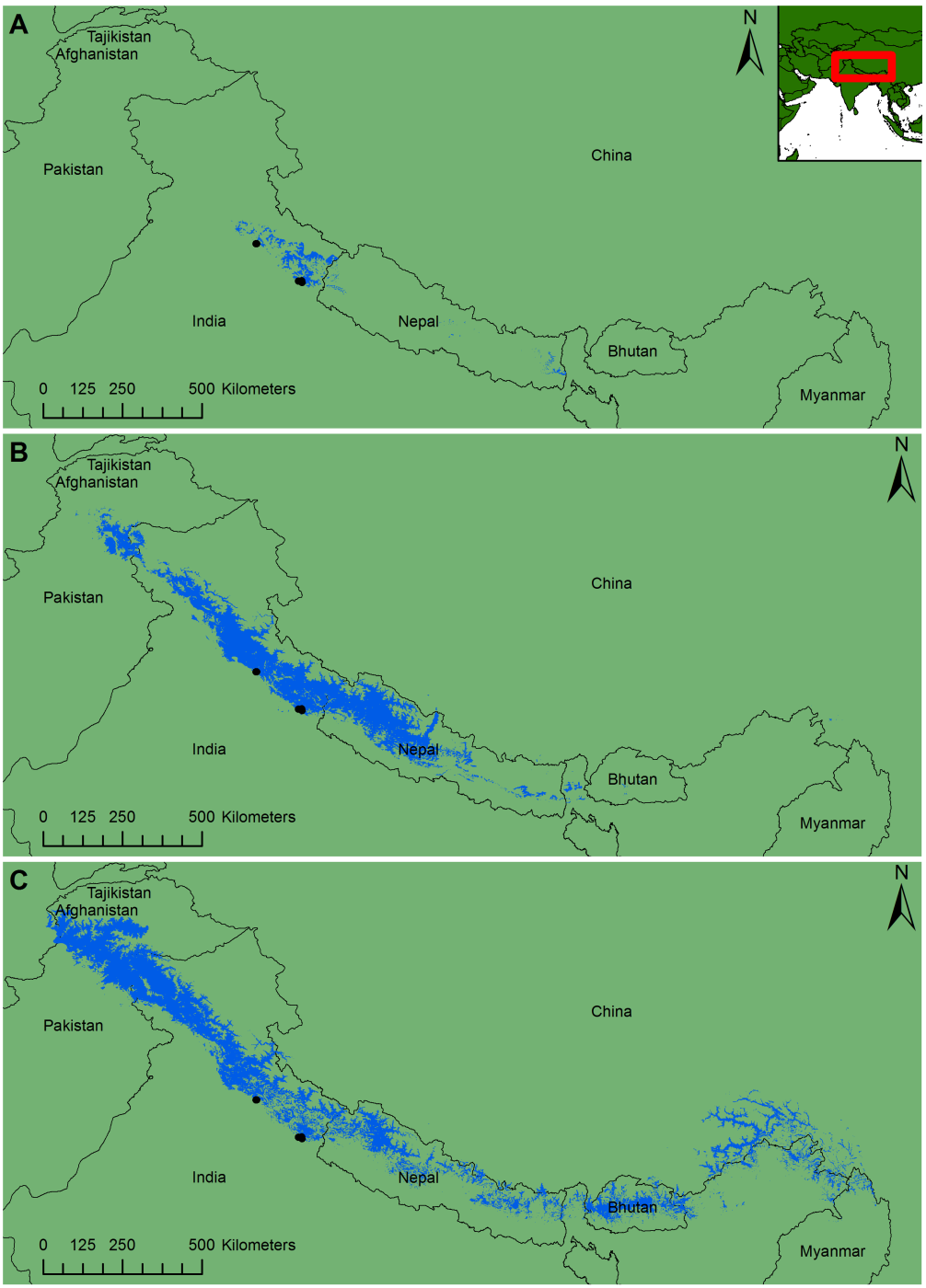
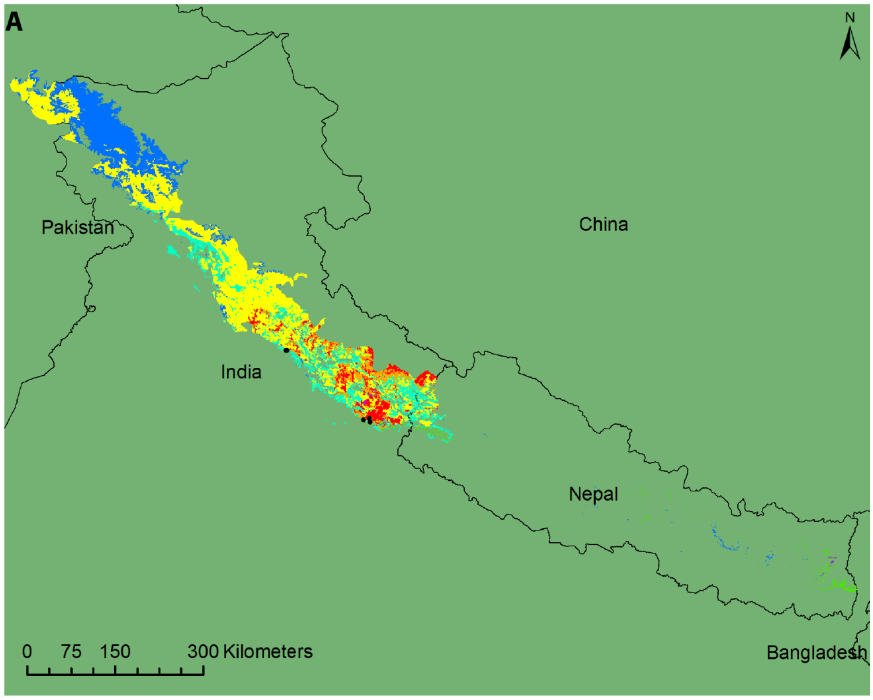


Figure 2.2: Species distribution maps. The plots show a climate map for: A) Himalayan quail (area = 9734 km^2) and species distribution maps for: B) cheer pheasant (area = 104228 km^2) and C) Himalayan monal (area = 162249 km^2). For all maps, black dots indicate Himalayan quail localities (NW = Mussoorie, SW = Nainital) and blue areas denote suitable habitat.



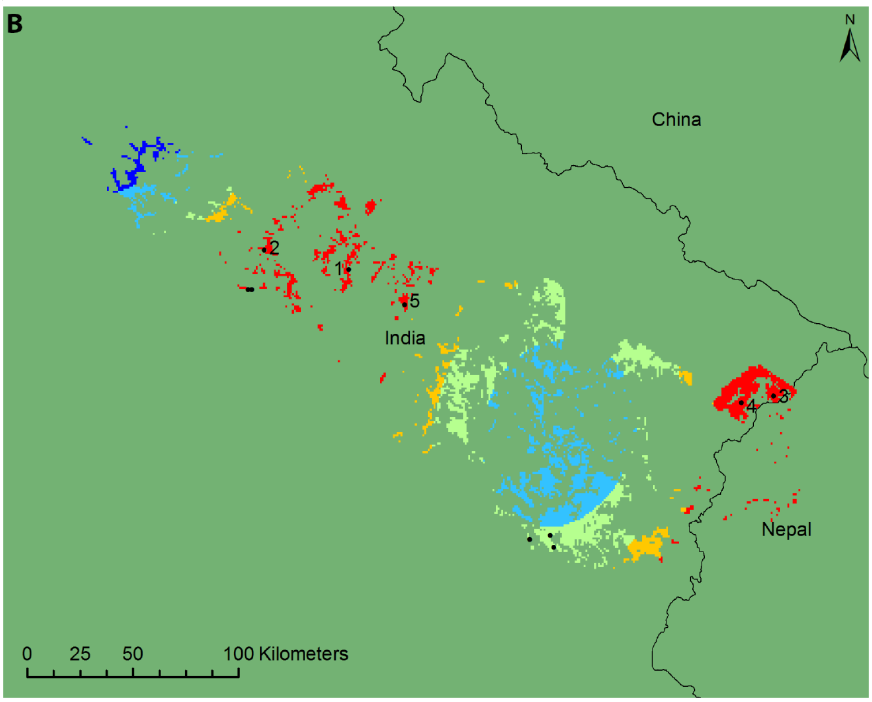


Figure 2.3: Maps showing overlap between Himalayan quail climate map and the Himalayan monal and cheer pheasant full species distribution maps. In A) red = three species overlap (4232 km^2), purple = overlap between the Himalayan quail and monal (159 km^2) and orange = overlap between Himalayan quail and cheer pheasant (4216 km^2). Yellow = overlap between monal and cheer (31696 km^2), blue = monal only (16336 km^2), cyan = cheer only (11314 km^2) and light green = Himalayan quail only (1056 km^2). Plot B) shows overlap between the Himalayan quail's climate map and full distribution models for the two proxy species as coloured by sampling effort. Warmer colours (red = 6-39 records/km^2 across an area of 923 km^2) indicate high priorities for resurveys (i.e. that have had a low sampling effort) and cool colours (dark blue = 143-209 records/km^2 across an area of 156 km^2) are low priorities (i.e. that have had a high sampling effort). Un-numbered black dots indicate Himalayan quail localities (NW = Mussoorie, SW = Nainital), whereas numbered black dots indicate priority re-survey sites (1 = Bhimleth, 2 = Khasonsi, 3 = Tyongi Pangu, 4 = Dug R.F., 5 = Chiribitiyakhal; see Table 2.2 for further details).

2.5 Discussion

The extinction status and distribution of the Himalayan quail are uncertain. Clarification of both is urgently needed to determine the most appropriate conservation action for the species. We found that the species may still be extant, albeit with considerable uncertainty surrounding the precise extinction date. This may be because the OLE technique assumes search effort never falls to zero for each annual time step, particularly around the time of extinction (Collen et al., 2010). In our study, this assumption is violated (see Figure 2.A1 in chapter appendix section 2.6.2), possibly extending the upper confidence interval of our extinction estimates. However, even if this is true, it is still important to search for the Himalayan quail to confirm extinction as the costs of incorrectly declaring extinction can be large. For example, giving up prematurely may doom the species to extinction (the 'Romeo error'; Collar, 1998) and re-appearances (the 'Lazarus effect'; Keith and Burgman, 2004) may waste conservation resources if costly and extensive surveys are undertaken.

Having established there is reason to consider the species as extant, and hence there is the possibility that it will be re-discovered, we need to identify the potential distribution. However, identifying such habitat through species distribution models is problematic due to data paucity and a temporal mismatch between reliable specimen records and the habitat covariates available for analysis. We identified the habitat requirements for cheer pheasant and Himalayan monal, which taken together should encapsulate those of the Himalayan quail. Our results suggest that areas other than the Indian localities of Mussoorie and Nainital should be searched and that Western Nepal appears less likely to contain suitable habitat for the Himalayan quail, although there is a large area of suitable habitat on the Indian side of the Nepalese border. We identified new areas which based on our models have a high potential occupancy likelihood and where intensive survey efforts have yet to be applied. There is an urgent need to better survey some parts of the Himalaya, a fact that is reinforced by the recent discovery of two new bird species in the Eastern Himalaya (WWF, 2009).

While proxy species have been used relatively extensively in conservation biology (Caro et al., 2005), they have not yet been used to potentially delimit areas of suitable habitat for rare species as far as the authors of this paper are aware. We consider the approach undertaken in this paper to be applicable for identifying macro-habitat preferences for other species, where data is limited and proxy species with similar ecological requirements and habitat use are available. While this method may be of use for identifying broad areas where species are likely to occur, it is unlikely that we can use the habitat preferences

of the cheer pheasant and Himalayan monal to provide information on the Himalayan quail's micro-habitat preferences. Even at the broadest scale caution must be used given Himalayan monal are altitudinal migrants: in the summer they appear on edges of alpine pastures near treeline, but in winter, are driven down by snow to mid-altitude forests (Del Hoyo, J., Elliott, A. and Sargatal, 2001). Conversely, the available evidence suggests that the Himalayan quail is not an altitudinal migrant (Das, 1995; Talwar, 1995). Thus, there could be a potential difference in habitat use in the winter, even if the Himalayan monal and Himalayan quail inhabit the same altitude. Similarly, cheer pheasant might occupy shorter, less thick grassland at the microhabitat scale than the Himalayan quail. As a result, it difficult to recommend discounting suitable areas that have been better-surveyed entirely, as previous searches might have been in the wrong microhabitats given the likely specificity of the Himalayan quail's habitat preferences.

In addition to these microhabitat considerations, there are also a number of specific methodological caveats for our analysis. First, it is possible that the Himalayan quail had a large historical distribution in the lowlands, but had been forced upwards into the mountains by historical habitat conversion (Rieger and Waltzhony, 1992). If so, our species distribution models might be incorrect, giving a false impression of the Himalayan quail's preferred climate envelope. However, the data we have do not support this hypothesis with no records of any kind from the lowlands. Second, it is highly likely that there is heterogeneity in detectability for each species depending on habitat. Investigating this further through surveys is important, as species occurrences and recorder biases could vary with covariates in unexpected ways (Yackulic et al., 2013).

A potential weakness in our approach was the possible effect of locational error from some of our point locality data on our modelling procedure. These locational errors could have made our models less accurate, both in terms of habitat associations and the resulting species distribution maps. In order to investigate this, we re-built our models based on a subset of data that all had the same locational error of 1 km^2. We found that the resulting AUC values remained high, suggesting that locational error had not compromised the accuracy of our models, which is reinforced by the fact that we did not use the Maxent default setting of 10,000 pseudo-absences in our modelling procedure. Similarly, given that the majority of records used in each of our models are accurate to 1 km^2 resolution and that the underlying land cover, topography and climate data were not at a particularly coarse resolution, we believe our approach to be robust.

An alternative approach to the one outlined in this paper could be to use a

rule-based assessment to identify potentially suitable habitat as has been used previously for both bird and mammal species (e.g. Hansen et al., 1995; Knick and Dyer, 1997). For example, we could identify the Himalayan quail's altitudinal range from suitable historical records and use descriptions of long grass being a preferred habitat to identify a subset of localities for searches. However, this approach may be difficult to implement and may provide biased maps. This is because grass is included as a broad habitat class in many typical land cover maps, such as the GLC2000 map (Bartholomé and Belward, 2005), making it difficult to identify the long grass the Himalayan quail is thought to prefer and allows for potential confusion with agricultural land (Fritz and See, 2008). As a result, we feel that our approach that makes use of NDVI data is likely to offer a better solution.

Our results indicate there is a large potential area of suitable habitat that has had little search effort applied to it, making ground-truthing some of the areas we have identified in our results the next logical step. To aid and encourage this, we include scalable .kml files of our results in the chapter appendix that identify potential areas for searching to enable field scientists and bird-watchers to plan effectively. As a starting point, we also include a list of five top priority localities with latitude and longitude information that should be surveyed first.

As our results indicate similarities in macro-habitat not micro-habitat, we recommend the following should be conducted prior to field surveys: 1) interviews with local herders, hunters, villagers and the State Forest Department staff, 2) a poster-mediated information campaign. Once this information is available, the exact nature of the field-surveys must account for local context and any searches for the Himalayan quail in suitable habitat must be undertaken with the collaboration and cooperation of local communities and landowners. Field surveys could potentially include the use of camera traps, trained dogs and audio-surveys/playback in order to try to find this cryptic and hard to flush species. We recognise that continued, unsuccessful searching for the Himalayan quail may eventually become cost-ineffective. Therefore, we suggest that the decision-theory framework provided by Chadès et al. (2008) could be used to determine how best to allocate limited conservation resources in the face of continued uncertainty (the authors are willing to provide analytical support). Once searches have been conducted, there is the potential to refine our species distribution maps using the resulting presence/absence survey data (a similar approach has been taken with Gurney's pitta (*Pitta gurneyi*) in Myanmar; Donald et al., 2009).

In conclusion, the new data, while imperfect, suggests the Himalayan quail may remain extant despite a lack of confirmed sightings for over 130 years. By using the habitat distributions of the cheer pheasant and Himalayan monal as proxies,

we identify new areas that could be used to guide search efforts on the macro-habitat scale, circumventing the minimal raw data available for the Himalayan quail. While robust, our maps are built on some more questionable assumptions, which require further testing. Overall, this is a novel approach to the problem of identifying survey areas for critically endangered species.

2.6 Appendix

2.6.1 Further details of historical records used in modelling procedures

The following tables provide further information on the locational error of the Himalayan quail (Table 2.A1), cheer pheasant (Table 2.A2) and Himalayan monal (Table 2.A3) records used in this Chapter. Additionally, further details on the observation type of the Himalayan quail records are provided (Table 2.A4).

	Pre-vetting		**Post-vetting**	
Locational error	**All**	**Post-1980**	**All**	**Post-1980**
Accurate (1 km)	13	1	12	1
Close (18.5km)	24	6	16	2
Vague (48 km)	8	6	4	4
Unknown	10	4	2	1

Table 2.A1: The geo-referencing accuracy of the Himalayan quail records in our database.

Locational error	**Number of records**	**Percentage of records**
Accurate (1 km)	255	55
Close (18.5 km)	89	19
Vague (48 km)	109	23
Unknown	16	3
Total	469	100

Table 2.A2: The number of post-1980 cheer pheasant records used in our Maxent models. The records are divided by geo-referencing accuracy.

Locational error	Number of records	Percentage of records
Accurate (1 km)	163	51
Close (18.5 km)	66	21
Vague (48 km)	78	25
Unknown	10	3
Total	317	100

Table 2.A3: The number of post-1980 Himalayan monal records used in our Maxent models. The records are divided by geo-referencing accuracy.

	Number of records	
Data	**Pre-vetting**	**Post-vetting**
All	55	34
Sight	29	17
Specimen	8	4
Unknown	15	13
Heard and Seen	1	0
Second hand	2	0

Table 2.A4: The number of Himalayan quail records before and after vetting.

2.6.2 Further details relating to Optimal Linear Estimation technique and results

Table 2.A5 shows the Himalayan quail data used in our extinction calculations and Figure 2.A1 shows the level of search effort near Mussoorie and Nainital.

Data used	Number of records	Number year classes	Extinction date	Upper CI	Lower CI
All	34	13	2023	2120	1999
Last five years	11	5	2010	2194	1996
Specimens only	4	2	N/A	N/A	N/A

Table 2.A5: Optimal Linear Estimation extinction dates based on vetted data. It was impossible to generate an extinction date based on specimen data only, so we present the results when both all records and records from the last five years were used in the calculations.

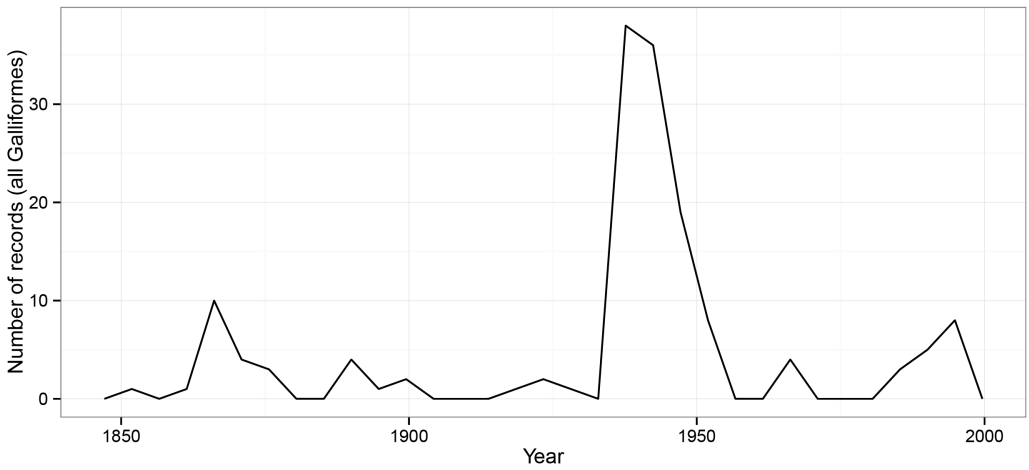


Figure 2.A1: Frequency polygon displaying search effort as measured by number of Galliformes records (all other species) in the locations (20 km buffer) of Mussoorie and Nainital. The troughs of this graph illustrate that search effort periodically drops to zero through time, thus violating one of the assumptions of the Optimal Linear Estimation method.

2.6.3 Further details relating to niche modelling procedures

The following tables provide further technical details relating to our niche modelling procedures.

Name	Species present
Baluchistan xeric woodlands	HM
Brahmaputra Valley semi-evergreen forests	HM
Eastern Himalayan alpine shrub and meadows	HM, CP
Eastern Himalayan broadleaf forests	HM
Eastern Himalayan subalpine conifer forests	HM, CP
Himalayan subtropical broadleaf forests	HM, CP
Himalayan subtropical pine forests	HM, HQ, CP
Northeastern Himalayan subalpine conifer forests	HM
Northwestern Himalayan alpine shrub and meadows	HM, CP
Northwestern thorn scrub forests	HM, CP
Nujiang Langcang Gorge alpine conifer and mixed forests	HM
Rock and Ice	HM, CP
Terai-Duar savanna and grasslands	CP
Upper Gangetic Plains moist deciduous forests	HM, HQ, CP
Western Himalayan alpine shrub and meadows	HM, CP
Western Himalayan broadleaf forests	HM, CP
Western Himalayan subalpine conifer forests	HM, CP

Table 2.A6: WWF Ecoregions used in Maxent analysis. Key: Himalayan quail = HQ, Himalayan monal = HM, cheer pheasant = CP.

Covariate	Species				
	Cheer ER	**Cheer SS**	**Hmonal all**	**Hmonal summer (lab)**	**Hmonal summer (part)**
Jan NDVI	1	1	0	0	0
Feb NDVI	0	0	1	1	0
Mar NDVI	0	0	0	1	0
Apr NDVI	0	1	0	0	0
May NDVI	0	0	0	0	0
Jun NDVI	1	1	1	0	1
Jul NDVI	0	0	1	0	0
Aug NDVI	0	0	1	1	1
Sep NDVI	0	0	0	0	0
Oct NDVI	0	0	0	0	0
Nov NDVI	0	0	0	0	0
Dec NDVI	1	1	0	0	0
Mean annual temperature	0	0	1	0	1
Mean annual variation temperature	1	1	1	1	1
Mean annual precipitation	0	0	0	0	0
Mean variation annual precipitation	0	1	1	1	0
Elevation	1	1	1	0	1
Slope	0	0	1	0	1
Aspect	0	0	0	0	0
Study site	0	1	1	1	1
Ecoregions	1	0	0	0	0
Occupied neighbour	0	0	0	0	0
Total number covariates	6	8	10	6	7

Table 2.A7: Details of covariates used in full environmental niche models for proxy species. Key: 1 = used, 0 = not used, ER = ecoregion, SS = study site, lab = labelled, part = partitioned by elevation. Note: ER and SS indicate different geographic extents used in the modelling procedure and lab and part indicate the different ways of delimiting the Himalayan monal's summer distribution.

Species	Model	Number unique records used	Study site delimitation method	Feature function	Regularisation value	Mean threshold value
Hquail	Climate	5	ON	l	5	0.715
Cheer ER	Full	192	ER	lqp	0.5	0.345
Cheer SS	Full	192	SS	lqp	1	0.305
Hmonal all	Full	216	SS	lqp	1	0.365
Hmonal summer (lab)	Full	10	SS	lp	1	0.548
Hmonal summer (part)	Full	39	SS	lp	0.5	0.308

Table 2.A8: Further details of our niche models. Note the number of unique records is lower than the number of records available. This is because Maxent automatically omits spatial duplicates of records. Feature function = the shape of the response curves for each model explanatory variable. Key: ER = ecoregions, SS = study site, lab = labelled, part = partitioned by elevation, ON = occupied neighbour, l = linear, q = quadratic, p = product.

Variable	Percent contribution	Permutation importance
Ecoregions	70.2	36.0
Annual mean variation in temperature	17.0	17.5
Annual mean variation in precipitation	3.8	4.1
Annual mean temperature	2.1	14.1
Elevation	2.0	23.0
Annual mean precipitation	1.7	3.2
Aspect	1.6	0.5
Slope	1.5	1.7

Table 2.A9: The relative importance of covariates used in the climate model for Cheer pheasant. Percent contribution is calculated as follows: in each iteration of the training algorithm, the increase in regularised gain is added to the contribution of the corresponding variable. Permutation importance is calculated as follows: for each variable in turn, the values of that variable on training presence and background data are randomly permutated. The model is then re-evaluated on the permuted data and the resulting drop in AUC is shown, normalised to a percentage.

Variable	Percent contribution	Permutation importance
Annual mean variation in precipitation	43.1	5.8
Elevation	22.1	33.6
Annual mean temperature	13.0	23.1
Annual mean precipitation	7.7	15.0
Annual mean variation in temperature	6.0	15.4
Slope	5.1	4.5
Aspect	3.0	2.7
Study site	0.0	0.0

Table 2.A10: The relative importance of covariates used in the climate model for Himalayan monal. For details on percent contribution and permutation importance see Table 2.A9.

Variable	Percent contribution	Permutation importance
Annual mean variation in temperature	66.8	91.9
Annual mean temperature	23.4	1.6
Elevation	9.0	4.4
Annual mean precipitation	0.8	2.1
Slope	0.0	0.0
Hquail mask	0.0	0.0
Annual mean variation in precipitation	0.0	0.0
Aspect	0.0	0.0

Table 2.A11: The relative importance of covariates used in the climate model for Himalayan quail. For details on percent contribution and permutation importance see Table 2.A9.

Variable	Percent contribution	Permutation importance
Ecoregions	59.2	51.2
Annual mean variation in temperature	15.9	25.3
Jun mean NDVI	11.1	2.6
Jan mean NDVI	5.5	8.9
Dec mean NDVI	5.2	6.9
Elevation	3.1	5.1

Table 2.A12: The relative importance of covariates used in the full niche model for cheer pheasant. For details on percent contribution and permutation importance see Table 2.A9.

Variable	Percent contribution	Permutation importance
Annual mean variation in precipitation	22.7	3.4
Jun mean NDVI	14.1	0.4
Jul mean NDVI	13.8	3.9
Elevation	10.1	9.3
Feb mean NDVI	9.5	33.0
Annual mean temperature	8.0	25.8
Annual mean variation in temperature	7.6	1.3
Slope	7.2	3.8
Aug mean NDVI	7.1	19.2
Study site hquail	0.0	0

Table 2.A13: The relative importance of covariates used in the full niche model for Himalayan monal. For details on percent contribution and permutation importance see Table 2.A9.

Chapter 3. Long-term relative range declines of Himalayan Galliformes show discordance with global Red List categories

3.1 Abstract

Identifying species declines is a key component of conservation biology and the IUCN Red List procedure. However, the Red List process is typically concerned with short time frames and global spatial scales, which means that long-term and sub-global declines may not accord with Red List categories. Given that Red List categories are often used for conservation baselines and priority-setting, we may need to examine species declines over longer time frames and sub-global spatial scales to make both informed and practical conservation decisions. Undertaking these analyses is challenging for a variety of reasons, which have restricted the number of opportunities for which this can be achieved. We use an assemblage of the highly threatened Galliformes taxon as a case study (N = 22 species with sufficient data) to examine long-term relative range declines and help focus conservation prioritisation in the Greater Himalayan region. We find that nine species have declined in relative geographic range size, seven have expanded and six have remained stable in the Himalayas. We also show that regional measures of range change are significantly positively associated with global measures of range change and identify common quail (*Coturnix coturnix*), Tibetan partridge (*Perdix hodgsoniae*) and Tibetan eared pheasant (*Crossoptilon harmani*) as having undergone the greatest relative declines within the Himalaya. We show that our global relative range decline measures fail to accord with contemporary global Red List categories, suggesting that for this suite of species historical relative declines do not accord with contemporary extinction risk. We discuss potential reasons for this discrepancy with implications for Galliformes conservation in the Himalaya.

3.2 Introduction

One of the overriding goals of conservation biology is to prevent species extinctions by targeting conservation resources to priority species (Vane-Wright et al., 1991). Thus, species that are at imminent risk of extinction require a higher need of protection (Possingham et al., 2002). The IUCN Red List

categorises species according to global extinction risk in a formal, quantitative assessment framework (IUCN, 2012). Species declines are a core part of the IUCN Red List assessment framework (Mace and Lande, 1991; Collar, 1996; Possingham et al., 2002). Thus, rapid species declines can lead to an enhanced risk of global extinction making it important to identify them and lessen the drivers of decline.

Although not expressly designed for such a purpose (Mace and Lande, 1991; Lamoreux et al., 2003; Rodrigues et al., 2006), the Red List is often used as a baseline for conservation priority-setting in practice (Schmeller et al., 2008; Brito et al., 2010; Martín-López et al., 2011). This means that species that have undergone rapid declines are often identified as priorities for conservation efforts by their Red List status (IUCN, 2012). However, the process of categorisation used to produce the Red List is very specific in how it reflects declines as a measure of extinction risk: namely declines are measured over short time frames and global spatial scales (details are discussed below). While this framework prevents imminent extinctions, sometimes we may need to examine species declines over longer time periods and sub-global spatial scales for a more informative picture of species conservation status to be made.

3.2.1 Species declines and temporal scale

The Red List assesses species declines over short time scales (three generations or ten years, whichever is longer; IUCN, 2012) and currently there are no official Red List provisions for examining species declines over longer time periods (Mace et al., 2008). Similarly, differences in the way Red List assessments have been made from 1964-1994 make it difficult to compare species declines over longer time periods (Mace and Lande, 1991). The rationale is that assessing declines over shorter time scales helps to identify rapid declines and prevent immediate extirpations. Yet this focus may only provide a snapshot of a species' conservation status and it can be informative to put declines in a proper historical context (Boakes et al., 2010). For example, slow, long-term species declines may not occur at rates that trigger a threat categorisation by the IUCN Red List, but it is still important to identify such declines because such species may require conservation action in the future (Wilson et al., 2011). Thus, pre-emptive approaches are recognised as being increasingly important to species conservation (Cardillo et al., 2004, 2006; Norris and Harper, 2004; Spring et al., 2007).

3.2.2 Species declines and spatial scale

The Red List threat categorisation process assesses species declines at global spatial scales because biodiversity and drivers of decline are largely independent of political boundaries (IUCN, 2012). Yet, in reality conservation planning and implementation is often targeted to sub-global spatial scales (Gärdenfors, 2001; Rodrigues and Gaston, 2002; Samways, 2003). Consequentially, it can be valuable to report national or regional species declines because if conservation action is targeted as a response to sub-global scale declines alone, our mitigation attempts may be ineffectual at preventing global extinction and may risk wasting scarce resources (Gärdenfors et al., 2001; Brito et al., 2010). This has been formally recognised by IUCN with guidelines available for regional and national Red List assessments (Gärdenfors et al., 2001; IUCN, 2003) whereby global assessments are made, then modified depending on the status of extra-regional or extra-national populations because such populations may influence the regional risk of extinction (Gärdenfors et al., 2001). Gärdenfors (2001) reports that there has been limited testing of its application and with a few notable exceptions (e.g. Eaton et al., 2005; Milner-Gulland et al., 2006; Pleguezuelos et al., 2010) this still appears to apply. Importantly, where detailed knowledge on regional and extra-regional populations is lacking, it can be difficult to implement IUCN regional Red List guidelines. Despite this obstacle, it is important to understand how best to target scarce conservation action across different spatial scales (Ferrier, 2002).

3.2.3 Challenges in examining species declines over both long time periods and sub-global scales

Examining species declines over both long time periods and sub-global spatial scales is challenging due to two related issues. First, the data available to identify species declines often precludes the usage of some of the most commonly used metrics of decline, such as measures of population size or geographic range. Second, although the Red List has official guidelines for sub-global Red List assessments, they are not designed to be used over longer time scales and can be difficult to implement without detailed knowledge of extra-regional populations of the focal species.

Species declines can be measured in a variety of ways depending on the data available (IUCN, 2012). Ideally, detailed population data should be used but this is frequently unavailable, so one commonly used proxy measure is to examine changes in geographic range size (Donald and Fuller, 1998; Shaffer et al., 1998; Telfer et al., 2002; van Swaay et al., 2008). However, it can even be difficult to use proxy measures such as range changes, because typically the

only data available are opportunistically collected and contain large spatio-temporal sampling biases (Boakes et al., 2010). Thus, it can be difficult to disentangle true range contractions from changes in sampling effort (Botts et al., 2012).

Correction factors such as Telfer's change index (Telfer et al., 2002) expressly look at measures of relative range change, rather than absolute changes and have been identified as appropriate for measuring species declines in the face of sampling biases (Boakes et al., 2010). Although the relative nature of Telfer's change index makes it impossible to identify range declines using more commonly used IUCN Red List metrics and to implement IUCN sub-global Red List assessments (i.e. due to a lack of population data), this is more than compensated for by our ability to identify long-term, negative changes in species conservation status independently from changes in sampling effort. Similarly, the degree to which a species' global range overlaps with a focal region (regional endemism) could provide a simple way of linking species declines to sub-global scales.

Using relative measures of geographic range change affords us an important opportunity to set conservation baselines and priorities over both long-time periods and sub-global scales, incorporating both global measures of relative decline and measures of regional endemism. Relative range change measures can also enable us to investigate how long-term, negative changes in species conservation status relate to contemporary assessments of extinction probability. This could help prevent further negative changes in species conservation status, species declines and extinctions by taking a more informed and pre-emptive approach than can be gained by using the Red List alone.

We investigate these issues further using a case study that focuses on a group of birds that live in a region of conservation concern and belong to an order where 25% of species are threatened with extinction. The Greater Himalayas provides important habitat for 24 species of the highly threatened Galliformes (gamebirds) bird order (ENVIS, 2007). These birds are found throughout the Greater Himalaya and exhibit a wide diversity of ecological characteristics, geographic distribution patterns and conservation statuses. Biodiversity in the Himalaya is thought to have undergone both large historical and contemporary declines (Bawa et al., 2007; Acharya et al., 2009; Xu et al., 2009), with increased localised extinction events and species declines predicted in the future (Chan et al., 2004; Brooks et al., 2006; Hoffmann et al., 2010). Knowing which Himalayan Galliform species have undergone long-term, negative changes in conservation status over longer time periods and at what spatial scale could better inform conservation baselines and priority setting within the region.

We aim to identify: 1) which Himalayan Galliformes species have undergone negative changes in conservation status the most both within the Himalaya and globally; and whether there are any differences between the two; 2) whether weighting regional relative declines by both global relative range declines and regional endemism (the proportion of the species global range contained within the Himalaya) changes our regional conservation priorities; 3) whether our measures of global and regional range changes are aligned with IUCN's Red List threat categories.

3.3 Methods

3.3.1 Galliform dataset

The GALLIFORM: Eurasian Database V.10 (Boakes et al., 2010, http://dryad.org) dataset includes 24 species of the avian order Galliformes found in, but not restricted to, the Greater Himalaya (for full details of the species included in the analysis and maps of the study region see Table 3.1 and Figure 3.A1 in chapter appendix section 3.6.1). Point locality data accurate to within 30 miles (48.24 km) were collected from a wide-range of sources including museum specimens, ringing records, biological atlas data and trip reports (for further details see Figure 3.A2 in chapter appendix section 3.6.1 and Boakes et al., 2010). The database contained 187,004 point locality records suitable for use in this study dating from 1625 – 2007 (for further details of the review process see chapter appendix section 3.6.1).

Absolute comparisons of geographic range changes are made difficult by biases in sampling effort and differences in sampling protocol (see chapter appendix section 3.6.1; Boakes et al., 2010). Thus, it is only possible to examine relative range changes (hereafter 'RRC'), which can be calculated by comparing geographic range sizes between two time periods for each species relative to the group as a whole (Telfer et al., 2002). The RRC technique has already been used successfully with this dataset to correct for sampling biases (albeit with the differing aim to examine geographic range dynamics; Mace et al., 2010). In addition, the technique makes few assumptions other than that each point locality record corresponds to some level of search effort in that particular year and location and is independent of sampling effort. In our dataset, the RRC scores for the 25 species of Galliformes in the Himalaya were not related to the number of records used to construct those RRC scores (spearman rank correlation, S = 1768, p = 1.00, r_s = 0.01).

3.3.2 Calculating the Telfer index

First, an RRC index was calculated for the Greater Himalaya. The point locality data were aggregated into a Behrmann equal area projection using a grid with cells measuring 48.24 x 48.24 km (see Figure 3.A3 in chapter appendix section 3.6.2), approximating to half a decimal degree resolution (the choice of grid cell size is further explained in chapter appendix section 3.6.2 in Table 3.A1). Only point locality data from Himalayan Galliform species were used to obtain measures of sampling effort in the Himalayan network of cells. Second, an RRC index was calculated for the globe using the same grid cell size as before (see Figure 3.A4 in chapter appendix section 3.6.2). While the global analysis examined range change for Himalayan species only, point locality data from other Galliform species (N = 94 species) were used in order to obtain a measure of sampling effort across the global network of cells. For both Himalayan and global analyses, data were split into pre- ($<$1980) and post-1980 ($\geq$1980) or early vs. later time bins to ensure congruence with other similar analyses conducted using this database (Mace et al., 2010) and to demarcate the rapid escalation of anthropogenic change starting around the year 1980 (Millennium Ecosystem Assessment, 2005). To control for change in geographical coverage with time, only cells that were surveyed for both time periods being compared were included.

The number of grid cells containing one or more records was counted for each species in each time period. Only species with a minimum of five cells in the early period were included in the analysis to avoid curvilinearity given that the rarest species have a far greater capacity to expand than decline (three species were excluded on these grounds; see Table 3.1; Telfer et al., 2002). These grid cells were then expressed as proportions of the total survey area and logit-transformed. A linear regression model was fitted to the logit-transformed proportions from the earlier and later periods and weighted to account for heteroscedasticity (for details see Table 3.A2 in chapter appendix section 3.6.2). Each species' standardised residual was then taken to represent an index of its change in geographic range size, relative to the trend in the whole group. Full details of the method are in Telfer et al. (2002). The standardised residuals represent relative change only and without calibration of the index it is difficult to know if the direction of the residuals represents absolute positive or negative changes.

3.3.3 Weighting regional range changes by regional endemism

The ranks from our global and regional RRC scores could be used naively to identify species with large regional relative declines relative to their global

relative declines as priorities for conservation efforts within the Himalaya. However, this only makes sense if we examine both scores as a function of the proportion of the focal species' global range contained within the Himalaya. Thus, regional range changes within the Himalaya were evaluated by examining: i) the global range changes for that species and ii) the degree of endemism to the Himalaya. The rationale is if a species has expanded within the Himalaya but declined in relative terms elsewhere, then the Himalaya might represent an important sanctuary for that species or vice-versa. This will be affected in turn by the proportion of a species' global range within the Himalaya; therefore species with large proportions of their global range within the study site will be most affected by range changes within the Himalaya and vice-versa.

Thus, the residuals from the linear model were extracted for both the Himalayan and global analyses and measures of endemism were calculated. The endemism measure reflected the proportion of each species' total global range that intersected with the Greater Himalaya. A continuous scale was used as definitions of endemism are inherently scale dependent (Laffan and Crisp, 2003) with 1 = 'endemic' and 0 = 'not endemic'. Each RRC score pertains both to the suite of species analysed and the network of total cells examined and are standardised as such. The modified score was calculated as follows: 1) the Himalayan RRC score was multiplied by the proportion of the species' global range that intersected the Himalaya and the global RRC was multiplied by the proportion of the species' global range outside of the Himalaya; 2) these two scores were then summed together. This had the effect of distinguishing between relative range changes in endemic vs. non-endemic species and between relative declines and expansions (see Tables 3.A3 to 3.A5 in chapter appendix section 3.6.3 for further details).

We present differences in range changes across both different geographic scales and weighted vs. un-weighted scores as differences in ranks.

3.3.4 Comparing conservation priorities identified by relative range declines to priorities identified by the IUCN Red List

We identified conservation priorities based on our relative range decline scores at different scales and compared them with IUCN Red List categories to assess concordance. Red List categories were taken from the IUCN Red List (IUCN, 2012) and converted to numerical scores (Least Concern = 1, Near Threatened = 2, Vulnerable = 3, Endangered = 4, Critically Endangered = 5). Thus, higher scores corresponded to a higher global extinction risk. As this was the opposite to our relative range declines scores (more negative scores corresponded to

larger relative declines), we reversed the polarity of the relative declines scores prior to analysis. Tied ranks were treated so each tied value was assigned the average of the ranks that would have been assigned without ties. Species omitted from the RRC analysis were included in all rankings apart from those based on RRC. Pairwise correlations were undertaken to investigate the relationships between different ranking methods. All analyses were carried out using R v.2.15.1 (R Development Core Team, 2012).

3.4 Results

3.4.1 Which Himalayan Galliformes species have declined in relative terms the most both within the Himalaya and globally; and are there any differences between the two?

The biggest relative regional declines were for Tibetan partridge (*Perdix hodgsoniae*), common quail (*Coturnix coturnix*) and Tibetan eared pheasant (*Crossoptilon harmani*), whereas the biggest relative declines globally were for Tibetan eared pheasant, hill partridge (*Arborophilia torqueola*) and rufous-throated partridge (*Arborophilia rufogularis*) (see Figure 3.1 and for a full breakdown see Tables 3.1 and 3.2).

A spearman rank correlation showed there was a significant positive association between regional and global RRC scores ($S = 609.34$, $r_s = 0.66$, $p<0.001$). While this does not reveal anything about the absolute rate of change due to the relative nature of the scores, it does suggest that if a species has declined in relative terms at the global scale, it is also likely to have declined in relative terms at the Himalayan scale. This implies that there is no difference in the pattern of relative declines and expansions between these two scales.

However, there were some clear outliers. In particular common quail and blood pheasant (*Ithaginis cruentus*) have worse relative regional range scores than global range scores (see Figure 3.2 panel A). Common quail and blood pheasant have both undergone global relative range increases but within the Himalaya itself, they have undergone relative declines. Whether these relative declines matter from a conservation perspective depends on the proportion of the species' global range contained within the Himalaya and is discussed in the following section.

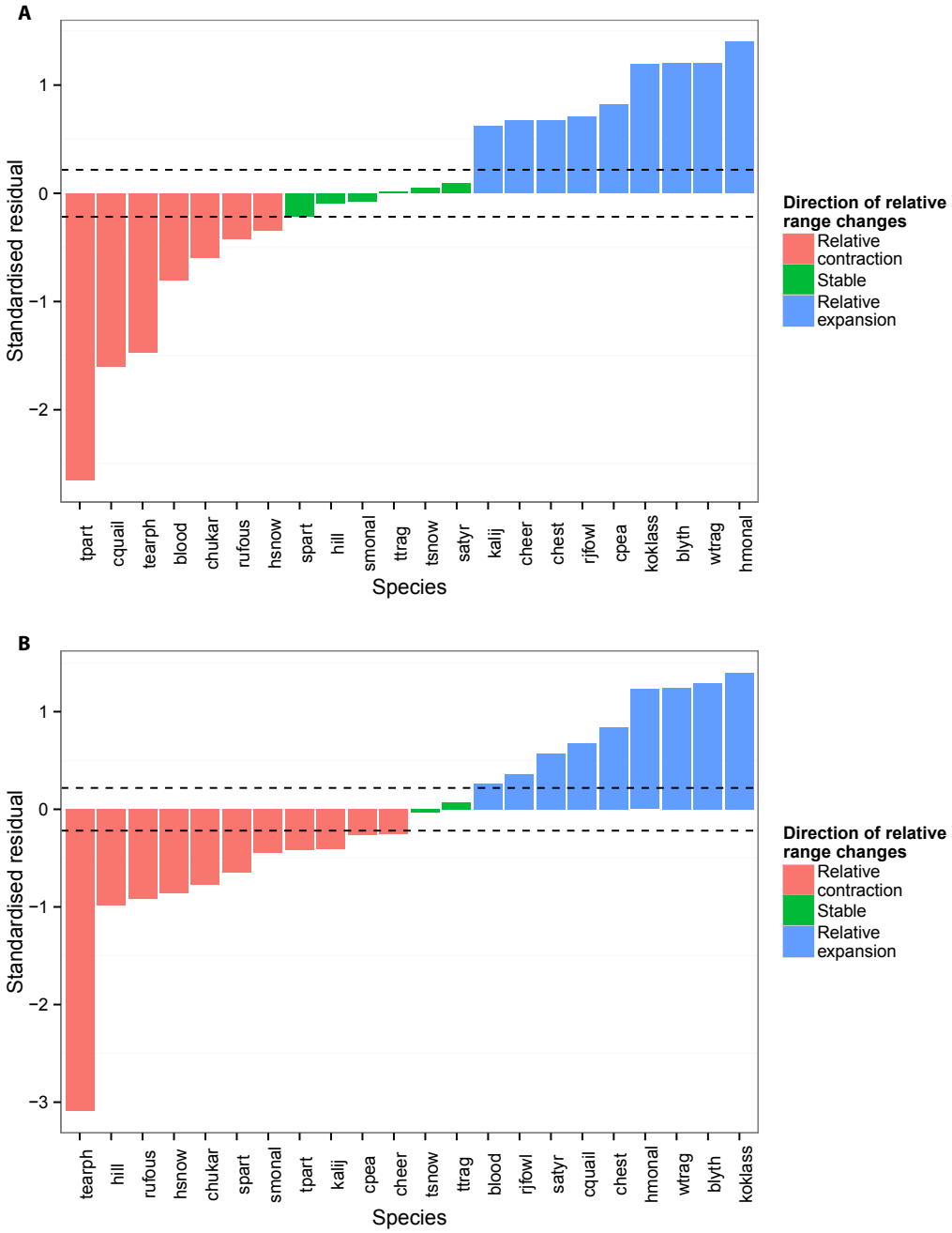


Figure 3.1: Bar graphs showing: A) regional relative range declines B) global relative range declines as measured by Telfer's change index. N = 22 species. Dashed lines indicate standard error bars (±1 SEM). Species codes are contained in Table 3.1. The colours of the bars correspond to the direction of the relative range changes: red = species with relative declines (standardised residuals more negative than the negative standard error bar), blue = species with relative expansions (standardised residuals more positive than the positive standard error bar) and green = stable species (standardised residuals are between the positive and negative error bars).

Common name	Binomial	Code	RL	Endemism	RRC_H	RRC_G	RRC_{WT}
Blood pheasant	*Ithaginis cruentus*	blood	1	0.22	-0.80	0.26	0.06
Blyth's tragopan	*Tragopan blythii*	blyth	3	0.23	1.20	1.29	1.28
Buff-throated partridge	*Tetraophasis szechenyii*	buff	1	0.21	NA	NA	NA
Cheer pheasant	*Catreus wallichi*	cheer	3	0.74	0.67	-0.25	0.43
Chestnut-breasted partridge	*Arborophila mandelli*	chest	3	1.00	0.67	0.84	0.65
Chukar	*Alectoris chukar*	chukar	1	0.03	-0.59	-0.77	-0.70
Common quail	*Coturnix coturnix*	cquail	1	0.04	-1.60	0.68	0.62
Hill partridge	*Arborophila torqueola*	hill	1	0.01	-0.09	-0.99	0.91
Himalayan monal	*Lophophorus impejanus*	hmonal	1	0.58	1.40	1.23	1.33
Himalayan quail	*Ophrysia superciliosa*	hquail	5	0.78	NA	NA	NA

Table 3.1: continued on next page

Common name	Binomial	Code	RL	Endemism	RRC_H	RRC_G	RRC_{WT}
Himalayan snowcock	*Tetraogallus himalayensis*	hsnow	1	1.00	-0.34	-0.86	-0.36
Indian peafowl	*Pavo cristatus*	cpea	1	0.12	0.82	-0.26	-0.08
Kalij pheasant	*Lophura leucomelanos*	kalij	1	0.33	0.62	-0.41	-0.03
Koklass pheasant	*Pucrasia macrolopha*	koklass	1	0.10	1.19	1.40	1.39
Red junglefowl	*Gallus gallus*	rjfowl	1	0.05	0.71	0.36	0.42
Rufous-throated partridge	*Arborophila rufogularis*	rufous	1	0.26	-0.42	-0.92	-0.74
Satyr tragopan	*Tragopan satyra*	satyr	2	1.00	0.09	0.57	0.07
Sclater's monal	*Lophophorus sclateri*	smonal	3	0.55	-0.07	-0.45	-0.22
Snow partridge	*Lerwa lerwa*	spart	1	0.40	-0.21	-0.65	-0.44
Temminck's tragopan	*Tragopan temminckii*	ttrag	1	0.06	0.01	0.07	0.11
Tibetan-eared pheasant	*Crossoptilon harmani*	tearph	2	0.62	-1.47	-3.09	-2.05
Tibetan partridge	*Perdix hodgsoniae*	tpart	1	0.17	-2.65	-0.42	-0.75

Table 3.1: continued on next page

Common name	Binomial	Code	RL	Endemism	RRC_H	RRC_G	RRC_{WT}
Tibetan snowcock	*Tetraogallus tibetanus*	tsnow	1	0.17	0.05	-0.03	0.02
Western tragopan	*Tragopan melanocephalus*	wtrag	3	0.78	1.20	1.24	1.20

Table 3.1: Raw and constructed scores used in analysis. RL = Red List category (Least Concern = 1, Near Threatened = 2, Vulnerable = 3, Endangered = 4, Critically Endangered = 5), RRC_H = Himalayan relative range change score, RRC_G = Global relative range change score, RRC_{WT} = Himalayan relative range change score weighted by endemism and global RRC score. More negative RRC scores indicate greater relative declines and higher conservation priorities whereas more positive RRC scores indicate greater relative expansions and lower conservation priorities. Note that the polarity of the weighted scores here were reversed prior to analysis to ensure priority ranks were congruent with each other. N = 24 species.

	Number of species	
Relative range change	**Region**	**Globe**
Contraction	9	9
Stable	6	2
Expansion	7	11

Table 3.2: A comparison of relative range change scores between two different spatial scales showing the breakdown in direction of relative range change and number of species. N = 22 species (we were not able to calculate range change estimates for Buff-throated partridge and Himalayan quail).

3.4.2 Does weighting regional RRC scores by regional endemism change our regional conservation priorities?

The biggest weighted regional relative declines were for Tibetan eared pheasant, hill partridge and Tibetan partridge (see Table 3.1). This suggests that the disparities seen between global and regional range changes for common quail and blood pheasant are less important once the proportion of the species' global range contained within the Himalaya is taken into account.

A spearman rank correlation showed there was a significant positive association between regional RRC scores and weighted regional RRC scores ($S = 488.28$, $r_s = 0.72$, $p<0.001$). Thus, accounting for the proportion of a species' global range contained within the Himalayas in this way, suggests that regional relative declines are similar to global relative declines at least in part due to similar drivers of range changes.

It is interesting to note that weighting regional RRC scores in this way suggests that common peafowl (*Pavo cristatus*), hill partridge and kalij pheasant (*Lophura leucomelanos*) are higher regional conservation priorities than would be suggested by the un-weighted regional RRC scores (see Figure 3.2). Similarly, common quail, blood pheasant and temminck's tragopan (*Tragopan temminckii*) are lower regional conservation priorities than they would first appear (see Figure 3.2 panel B).

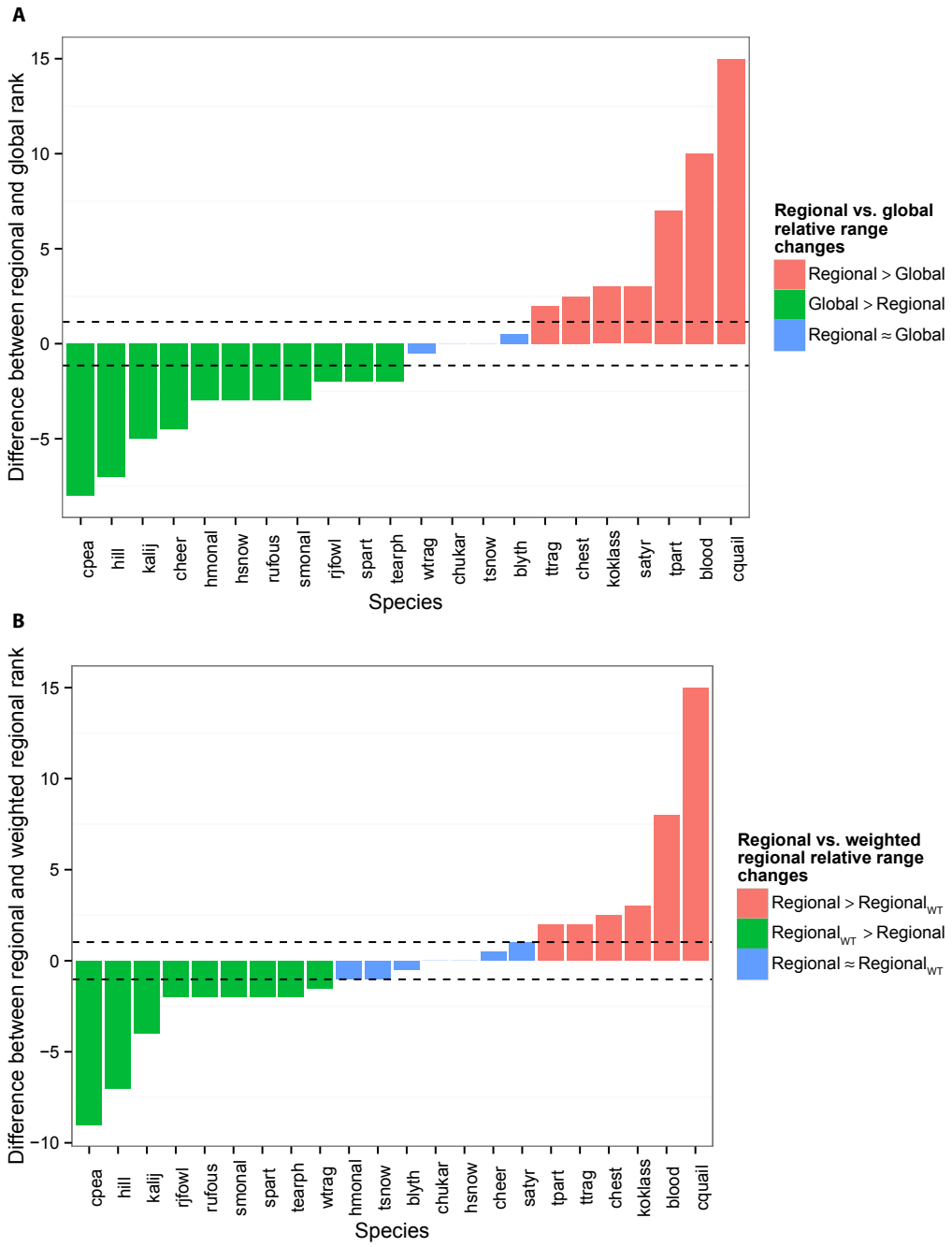


Figure 3.2: Bar graphs showing: A) differences in ranks between regional vs. global relative range changes; B) differences in ranks between regional vs. weighted($_{WT}$) regional range changes. N = 22 species. In A), positive differences (denoted by red bars that are more positive than the positive error bar) indicate that the regional situation is worse than the global situation. This could be due to a regional contraction and global expansion, large regional contraction and small global contraction or a small regional expansion and a large global expansion. Negative differences (denoted by green bars that are more negative than the negative error bar) indicate that the regional situation is better than the global situation. This could be due to a regional expansion and global contraction, small regional contraction and large global contraction or a large regional expansion and small global expansion. In B), positive differences (denoted by red bars that are more positive than the positive error bar) show where weighting regional relative range changes by regional endemism, lowers their regional priority ranking. Negative differences (denoted by green bars that are more negative than the negative error bar) show where the regional priority ranking of regional relative range changes is actually increased when weighted by regional endemism. Dashed lines indicate standard error bars (±1 SEM). Species codes are contained in Table 3.1.

3.4.3 Are our measures of global and regional range changes aligned with IUCN's Red List threat categories?

A spearman rank correlation showed there was a non-significant negative association between regional RRC scores weighted by global relative declines and Himalayan endemism and IUCN Red List category (S = 2272.39, r_s = -0.28, p = 0.20). Additionally, there was a non-significant, negative association between raw regional RRC scores and Red List category (S = 243.72, r_s = -0.32, p = 0.14) and a non-significant, negative association between global RRC scores and Red List category (S = 2217.50, r_s = -0.25, p = 0.27). Note that RRC scores were reversed in polarity before this analysis to priorities given by the RRC scores were of the same polarity as the priorities given by Red List category scores (i.e. whereas the most negative raw scores were originally the top priorities, the most positive modified scores were the new top priorities). Thus, priority ranks from the Red List category scores were also those with the most positive values (i.e. species with a high global extinction risk). Thus, if our correlations were significant, it would have indicated that species that had undergone large, long-term relative declines were less likely to have a higher risk of global extinction according to their Red List category.

3.5 Discussion

Identifying species declines is important because it allows conservationists to both provide priority action through IUCN classification (IUCN, 2012) and to determine whether there are any common factors placing certain taxa or areas at risk (Kotze and O'Hara, 2003; Eken et al., 2004; Sodhi et al., 2008). However, a variety of constraints have restricted the opportunities for this to be achieved over long time frames and sub-global spatial scales. We examined long-term relative declines of Galliform species at both global and Himalayan scales using a relative index of range change to assess how spatial scale related to our regional conservation priorities and whether historical relative declines accorded with contemporary Red List categories. We found that: 1) if Himalayan Galliform species have declined in relative terms globally, they were also likely to have declined in relative terms regionally; 2) weighting regional relative declines by global relative declines and endemism to the Himalaya did not change the resulting regional priorities and 3) none of our measures of long-term relative decline were linked to contemporary IUCN Red List categories regardless of spatial scale.

It is challenging to examine long-term species declines using opportunistically collected data and at sub-global spatial scales using conventional Red List metrics and methodologies (i.e. Regional and National Red List assessment

protocols). Nevertheless, setting species declines in a broader spatio-temporal context could better inform us when setting conservation priorities, allowing for a more pre-emptive approach to preventing extinctions. Using Telfer's relative index of change (Telfer et al., 2002) with measures of geographic range allows us to do this and by disentangling true relative range changes from biases in sampling effort, is the most appropriate way of using opportunistically collected data to examine relative species declines over different temporal and spatial scales. For Galliform species in the Himalaya this is invaluable, given that the Himalaya has been identified as an important conservation region with significant future biodiversity losses predicted (Klein et al., 2004; Walker et al., 2006). For example, Pandit et al. (2006) reported that in the Indian Himalaya alone, current levels of deforestation are likely to wipe out almost a quarter of endemic species, including 366 endemic vascular plant taxa and 35 endemic vertebrate taxa by 2100.

We found that there were similarities in the priorities given by regional and global relative range declines for Himalayan Galliformes. However it is not obvious why this should be the case: for example, the pattern we show could be due to similarities between geographic range sizes across the two scales, similar drivers of relative declines over the same spatial scale, or even different drivers of relative decline operating over the same temporal scale. Further investigation into the nature of these drivers of long-term range changes is needed if we are to effectively ameliorate them or understand how to best mitigate for them.

If we are to use regional relative range declines to set ecologically meaningful regional conservation baselines and priorities within the Himalaya, we must also consider global relative declines and measures of regional endemism. Our results showed that when we do this, the resulting priority ranks are similar to those based on raw regional RRC scores alone. Practically, this implies that special provisions do not need to be made for the regional conservation of Galliformes within the Himalaya in a broad sense, although important outliers exist. In the absence of further knowledge, we might argue that basing conservation baselines and priorities on regional RRC scores for this region and suite of species is largely unnecessary and that global RRC scores could be used on their own to form part of the regional conservation decision-making processes.

Given that a species' IUCN Red List category is commonly used to inform conservation priority-setting, it is important to assess the accordance between our measures of long-term changes in species conservation status and Red List category. We expected that our priorities based on global Red List categories would be broadly similar to those from our global RRC scores (especially given that we do not examine the relative scores themselves but ranks), because

species declines form a core part of the IUCN Red List assessment protocol (IUCN, 2012). However, our results suggested that there is no statistically significant accordance between Red List categories and long-term relative declines as measured by the Telfer method (Telfer et al., 2002) either at global or sub-global levels. This discrepancy could be due to a variety of reasons: the first is that the Red List is an aggregate score that is based on measures of rarity as well as declines (Telfer et al., 2002), which is supported by our correlation matrix that shows Red List threat category increases with the degree of endemism to the Himalaya (see Table 3.A6 in chapter appendix section 3.6.3). If this is the case, the next step would be to investigate whether the relative declines that we observe differ for species contingent on the size of their historical range. The second reason is that given that the Red List examines a snapshot of a species' trajectory to extinction it could be that the shapes of such trajectories have since changed over contemporary timescales. Future work could concentrate on assessing range changes over a finer temporal resolution to investigate this finding further.

An important consequence of the lack of accordance between the Red List and our relative range decline measures, is that if we use Red List categories to set baselines for regional conservation prioritisation in the Himalaya, we are likely to undervalue species that are categorised as non-threatened ('Least Concern' or 'Near Threatened') but have undergone large historical relative declines. Thus, there are a few species that we feel should be better considered in regional conservation action plans, such as Tibetan partridge and Tibetan-eared pheasant. While these species are not listed as threatened by the IUCN Red List, their geographic ranges have declined in relative terms considerably within the Himalaya, which comprises a major part of their global range and over a long time period. Given that we know that deforestation and hunting are likely to be important threats for these species (Lu and Zheng, 2007), it is possible that we have thus far underestimated their relative declines, especially if the intensity and severity of these threats have remained high throughout time.

Our results come with some caveats: the first is to note that increased recorder attention towards threatened species might bias our assessments of range change (Telfer et al., 2002). Thus, the reason we found non-threatened species to have undergone the greatest relative range declines could be because they have received less recorder attention in the later time period. However, it is unlikely that this is true in our study as we found the proportion of records from threatened species approximately equal to those of non-threatened species between pre- and post-1980 time bins (see chapter appendix section 3.6.1). The second caveat is that our attempt to link spatial scale to

species declines may be overly simplistic; thus, even if a large proportion of a species' total range is found within a focal region, that part of the range may not contain breeding populations and could even act as a population sink (c.f. IUCN Regional Red List assessment protocol). We recognise this limitation but emphasise that it is difficult to account for this in our approach in absence of population data. The final caveat is that other considerations for species priority setting should be examined in addition to range declines and Red List categories. Our study has focused on declines and changes in conservation status, but we recognise that factors such as human values (Nunes and van den Bergh, 2001), biodiversity benefits, probability of success (Kennedy et al., 1996) and economic costs (Naidoo et al., 2006) may also be accounted for before any specific conservation action is taken.

In conclusion, we showed that apart from a few exceptions, global range changes were similar to regional range changes for Himalayan Galliformes. While our relative range change scores did not relate to IUCN Red List categories in a clear way, this does not preclude us from using them to help set regional conservation priorities. We suggest relative range change scores can be used to provide useful insights for conservation priority-setting across a range of spatio-temporal scales where detailed population data is lacking. The challenge in the future is to: 1) explore the exact nature of the drivers of relative range declines in Himalayan Galliforms; 2) examine relative declines over finer temporal resolutions; 3) explore techniques to convert relative range change scores to measures of absolute range change to gain a better understanding of the ramifications of such changes.

3.6 Appendix

3.6.1 Further details of the study region and database used

The numbers of records within our database were quantified for each Himalayan Galliform species. Himalayan species were those found in but not restricted to the Greater Himalayas (see Figure 3.A1). Any records without latitude/longitude co-ordinates or a collection date were removed. The data subsequently went through an extensive review process with Jon Dunn and Dr Philip McGowan to remove any manifestly wrong records (where the species had been introduced or the record had been incorrectly geo-referenced and the overall data quality was assessed). We did not remove records that we thought were wrong that may have been due to distributional differences between early and late ranges. We sought to exclude spurious/highly improbable records only e.g. records of endemic Himalayan species occurring in localities such as Florida.

Figure 3.A1: Map displaying the global location of the Greater Himalaya, which covers approximately 700,000 km^2 as delimited by WWF Ecoregions (Wikramanayake et al., 2002b).

We found that there was a greater recording effort in the later time period (see Figure 3.A2) as shown by two-tailed chi squared tests (Himalaya: number years pre-1980 = 354, number years post-1980 = 27, number records pre-1980 = 228, number records post-1980 = 2237, records/year pre-1980 = 6.46, records/year post-1980 = 82.55, χ^2 = 65.05, df = 1, p<0.001; Globe: number years pre-1980 = 354, number years post-1980 = 27, number records pre-1980 = 110621, number records post-1980 = 76383, records/year pre-1980 = 312.44, records/year post-1980 = 2829.00, χ^2 = 2015.98, df = 1, p<0.001), justifying our use of the Telfer correction factor technique to examine range changes. Additionally, one-tailed two-sample tests for equality of proportions with continuity correction showed that there was a significant increase in the proportion of records from threatened species between pre- (proportion = 8.8 x 10^{-4}) and post-1980 (proportion = 1.9 x 10^{-2}) bins for the Himalaya (χ^2 = 35.37, df = 1, p<0.001). Yet, when adjusted to account for the unequal time bins, there was no significant difference between pre- (proportion = 0.31) and post-1980 (proportion = 0.50) (χ^2 = 0.27, df = 1, p>0.1), suggesting in our focal region there had not been a significant bias in record collection towards threatened species.

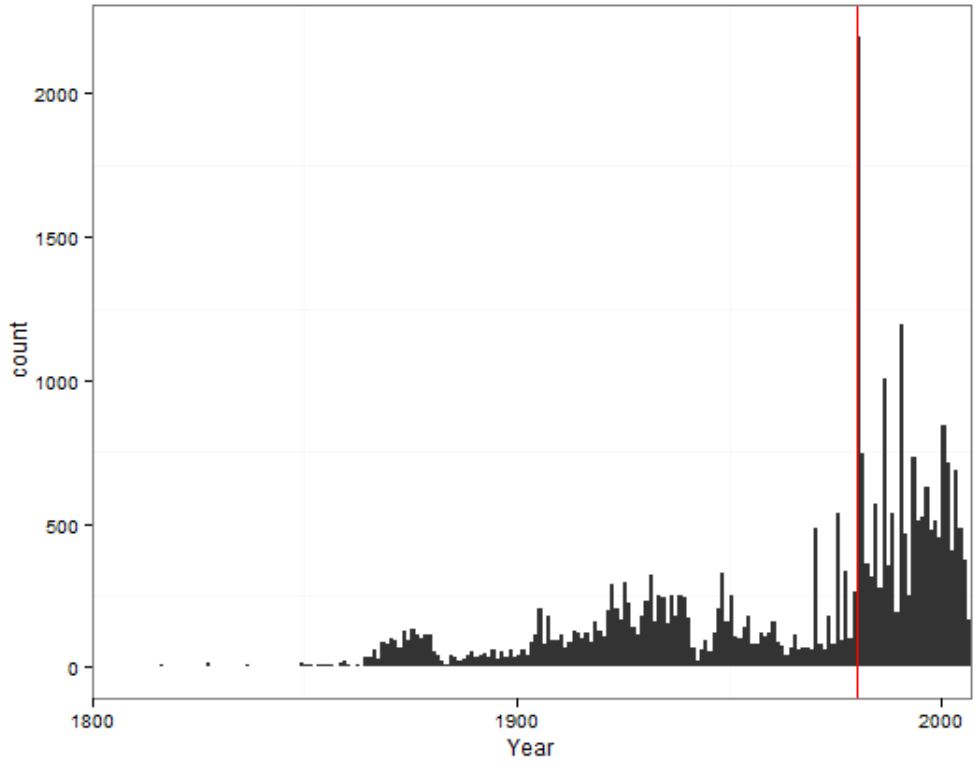


Figure 3.A2: Distribution of global records for Himalayan species between pre- and post-1980 bins. The red line denotes the year 1980. 187,004 individual records were used for global analysis with 4525 individual records used for the Greater Himalaya.

3.6.2 Further details on calculating the Relative Range Change (RRC) index

The two grid-based networks that were used in the creation of our RRC index are shown in Figures 3.A3 and 3.A4.

Scaling issues were thought to be important when constructing our networks of cells that had been sampled from both time periods, given that the larger the grid cell, the greater the theoretical likelihood that a record would be included in the locality. At the same time, it was thought it would be more difficult to detect localised extinction/colonisation, so we investigated this trade-off further by experimenting with three different cell sizes: 0.1, 0.25 and 0.5 decimal degrees (see Table 3.A1).

We found that as cell size increased so did aggregation i.e. the probability that a point was included in the same cell as its neighbour increased. Furthermore, with increased cell size the overall number of cells in a network decreased, which could possibly have made it harder for locally range restricted species to have fulfilled the minimum criteria of 5 cells in the early period; however, it was found that at the largest cell size this was not the case (see Table 3.A1). With increased cell size, the total proportion of potential cells acting as part of the network increased, indicating a larger proportion of the study site was

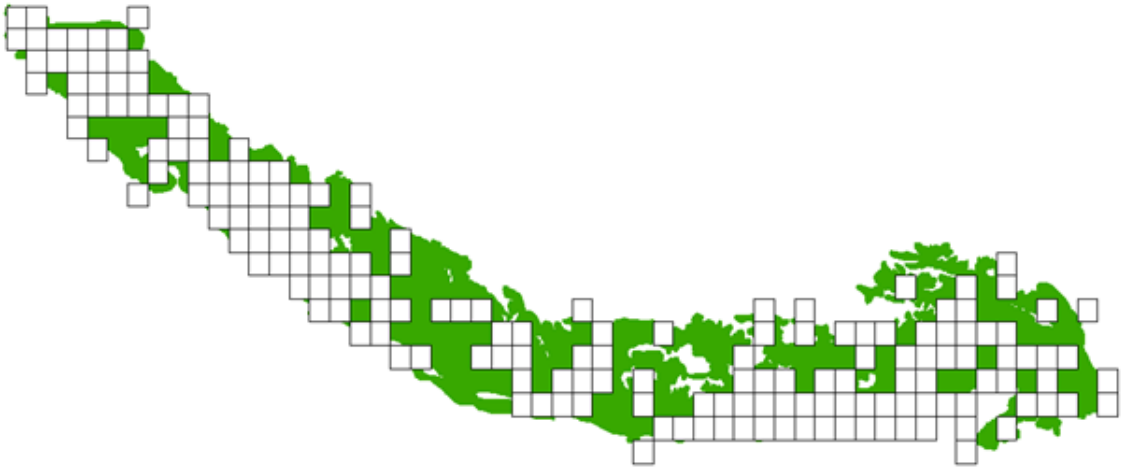


Figure 3.A3: Greater Himalaya showing study site RRC network. Study site grid cell size = approximately 50 x 50 km. Number of cells in network = 165.

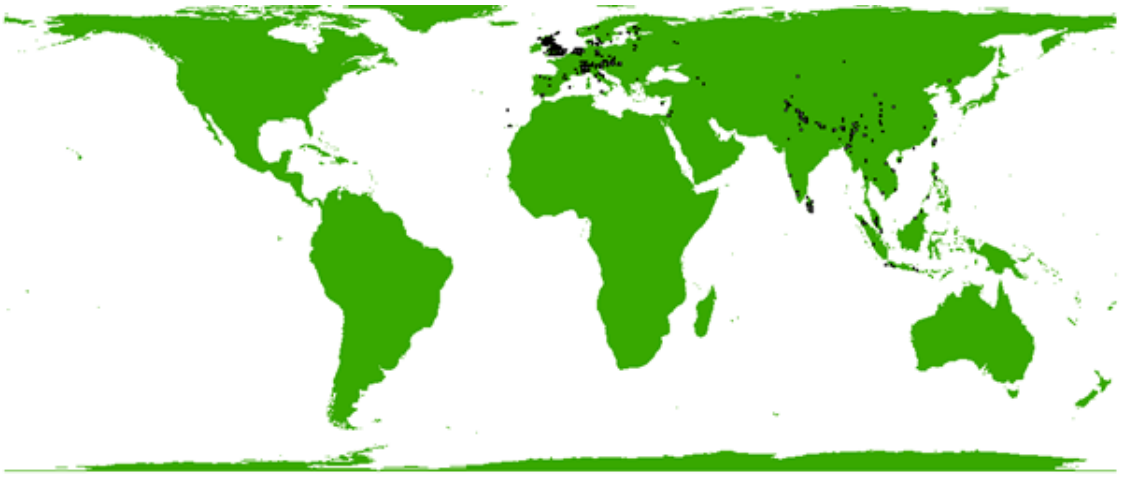


Figure 3.A4: World map showing global RRC network. Grid cell size = approximately 50 x 50 km at equator. Number of cells in network = 428.

included in the network. Therefore, in general, we found it preferable to choose as large a cell size as possible to include as many locality points and grid cells in the cell network as we could, thereby increasing the proportion of the study site contained within the cell network (see Table 3.A1). The drawbacks of using a larger cell size included not obtaining sufficient cells in the early period for some species and potentially missing changes in occupancy, particularly for range-restricted species. We conducted our analyses using both 0.5 decimal degree and 0.25 decimal degree scales (but not the 0.1 decimal degree scale). There were no differences in our final results, so we were justified in using the largest 0.5 decimal degree scale.

The weighted linear models used to create our RRC index are presented below in Table 3.A2.

Scale	% points in network	# cells in network	% ss cells in network	% spp excluded
10″	70.8	564	17.8	4
0.25′	89.4	452	36.6	8
0.5′	91.8	165	50.4	8

Table 3.A1: The effect of grain size on the network of cells included in the RRC analysis. The largest cell size has more points in the network, has fewer cells in the network but covers more of the study site and only 8% of species are excluded.

Network	Un-weighted regression equation	Weighted regression equation	No. iterations required
Global	y=0.91x-0.14, r^2 = 0.93	y=0.90x-0.17, r^2 = 0.91	6
Region	y=0.81x-0.38, r^2 = 0.42	y=0.76x-0.49, r^2 = 0.39	3

Table 3.A2: The regression equations from the RRC analyses of two different networks.

3.6.3 Further details on how we assessed both regional and global patterns of relative range declines

We devised a system to give a higher conservation priority ranking to species' relative range declines when weighted by the proportion of that focal species' global range contained within the Himalaya (as outlined in the methods section). We tested whether our methodology gave sensible conservation priorities a priori as seen below in Tables 3.A3 to 3.A5. This was done by using dummy data to represent RRC scores and measures of endemism. Thus, the results shown below show the final rankings and not raw scores.

	$E_{H,L}$	$E_{H,M}$	$E_{H,S}$	$D_{H,L}$	$D_{H,M}$	$D_{H,S}$
$E_{G,L}$	1	2	3	5	6	8
$E_{G,M}$	2	4	5	7	8	10
$E_{G,S}$	3	5	6	8	9	11
$D_{G,L}$	5	7	8	10	11	13
$D_{G,M}$	6	8	9	11	12	14
$D_{G,S}$	8	10	11	13	14	15

Table 3.A3: Theoretical scenarios that reflect our a priori expectations of regional conservation priorities based on range relative declines alone if endemism to the Himalaya is ignored. High scores represent top priorities; 'E' = relative range expansions; 'D' = relative range declines; $_{H,G}$ = Himalaya or Globe; $_{L,M,S}$ = large or medium or small.

We also investigated whether priorities based on our various measures of relative decline were related to each other, the IUCN Red List and the

	$E_{H,L}$	$E_{H,M}$	$E_{H,S}$	$D_{H,L}$	$D_{H,M}$	$D_{H,S}$
$E_{G,L}$	1	7	13	19	25	31
$E_{G,M}$	2	8	14	20	26	32
$E_{G,S}$	3	9	15	21	27	33
$D_{G,L}$	4	10	16	22	28	34
$D_{G,M}$	5	11	17	23	29	35
$D_{G,S}$	6	12	18	24	30	36

Table 3.A4: The effect on regional conservation priorities by incorporating a measure of endemism to the Himalaya where endemism is high. High scores represent top priorities; 'E' = relative range expansions; 'D' = relative range declines; $_{H,G}$ = Himalaya or Globe; $_{L,M,S}$ = large or medium or small.

	$E_{H,L}$	$E_{H,M}$	$E_{H,S}$	$D_{H,L}$	$D_{H,M}$	$D_{H,S}$
$E_{G,L}$	1	2	3	4	5	6
$E_{G,M}$	7	8	9	10	11	12
$E_{G,S}$	13	14	15	16	17	18
$D_{G,L}$	19	20	21	22	23	24
$D_{G,M}$	25	26	27	28	29	30
$D_{G,S}$	31	32	33	34	35	36

Table 3.A5: The effect on regional conservation priorities by incorporating a measure of endemism to the Himalaya where endemism is low. High scores represent top priorities; 'E' = relative range expansions; 'D' = relative range declines; $_{H,G}$ = Himalaya or Globe; $_{L,M,S}$ = large or medium or small.

proportion of each species' global range contained within the Himalaya as seen in Table 3.A6.

	RL	**Endemism**	**RRC_H**	**RRC_G**
RL	*****	0.615	0.323	0.252
Endemism	0.002	*****	0.212	0.068
RRC_H	0.142	0.343	*****	0.656
RRC_G	0.258	0.762	0.001	*****

Table 3.A6: Inter-correlation matrix of the raw scores (spearman rank). Key: the upper diagonal part contains correlation co-efficient estimates and the lower diagonal part shows p-values. There is a significant positive association between Red List category and Endemism to the Himalaya but also Himalayan and global relative range declines. Thus, endemic species are likely to be globally threatened. Similarly, species that have undergone relative regional declines are also more likely have to undergone relative global declines. Key: RL = Red List, RRC_H = relative range changes at the Himalayan scale and RRC_G = relative range changes at the global scale.

Chapter 4. Are widespread species that have undergone long-term range contraction overlooked in global assessments of likelihood of extinction?

4.1 Abstract

Geographically widespread species play an important role on our planet driving both biodiversity patterns and ecosystem function but concerns exist that we are overlooking long-term pervasive declines in such species. Conservation biology is generally concerned with narrowly distributed species, as a small range size has been shown to be the biggest single predictor of extinction and it is easier to monitor and track the status of such species over small time periods. Thus, adjustments to the ways we assess species declines in widespread species could be needed in order to provide a more informed approach to their conservation. We examine long-term range changes to inform the way we assess such changes in the present. To do this, we use a comprehensive database of historical Galliformes records from 94 species comprising over 161,000 records covering the past 350 years to show that there are no differences in historical range declines between restricted-range species and widespread species (using two different methods). We also find that across species that have declined, widespread and restricted species (<50,000 km^2) are equally likely to be listed as threatened by contemporary IUCN Red List assessments. This suggests implicit support for the Red List framework with no special adjustments to Red List procedures needing to be made when assessing range declines in widespread species.

4.2 Introduction

The current human-driven declines in global biodiversity (Secretariat of the Convention on Biological Diversity, 2010) have led to significant international policy responses through a range of multilateral environmental agreements (e.g. CBD, 2010). Many of these agreements rely on assessments of change in the conservation status of individual species. The scale of recent declines in species' conservation status has reinforced the importance of quantifying and documenting such declines (Tingley and Beissinger, 2009). Change in species conservation status, or likelihood of extinction are formally

accounted for in global and national assessments by IUCN Red List methodology and are reflected in Red List categories of global extinction risk (IUCN, 2012). In order to quantify such changes in conservation status, data must be collected to obtain either direct estimates or proxies of population size from different time periods (e.g. Donald and Fuller, 1998; Mace et al., 2010; Botts et al., 2012). This data can be obtained from many sources such as monitoring and bird atlases, particularly for recent time periods (e.g. Robbins et al., 1989; Buckley et al., 2014).

Detecting change in conservation status cannot be undertaken for all species as resources are limited. Furthermore, the spatial and temporal scale at which change is being assessed will also influence how feasible it is to gather appropriate data to make meaningful and defensible assessments: i.e. it is harder to monitor populations, or their proxies, across a larger geographic area than it is over a smaller area and over longer time periods. In other words, as spatial and temporal scale increase it is more difficult to assemble data that would allow the detection of changes in conservation status (for a more comprehensive discussion of these issues, see thesis Chapter 3).

The global standard for assessing the probability of extinction status is the IUCN Red List and this uses a variety of criteria to assign species to categories denoting how likely they are to go extinct (IUCN, 2012). The size of a species' geographic range is one of the most widely used criteria and 47% of the 4440 threatened species (mammals, birds, amphibians and gymnosperms) on the Red List have been listed on the basis of range criteria alone (Gaston and Fuller, 2009). Thus, species that possess small ranges are thought to be at an enhanced risk of extinction, requiring immediate conservation attention (Simberloff, 1986; Caughley, 1994; Sodhi et al., 2008). In addition, IUCN's methodology for assessing population declines operates over short time scales (three generations or ten years depending on whichever is longer; IUCN, 2012). Both of these issues, short time scales and the additional emphasis on species with small geographical range, may lead to widespread species that have had marked and sustained declines over a long time period not being listed as threatened. This effect could explain the results of a previous study (see thesis Chapter 3), which found Red List categories did not accord with long-term range changes in Himalayan Galliformes (gamebirds).

Failure to track marked and sustained declines in widespread species is likely to have important consequences: first, many species that were historically widespread and common have become threatened or gone extinct often without first becoming threatened with imminent extinction (Gaston and Fuller, 2008) e.g. passenger pigeon (*Ectopistes migratorius*; Gaston, 2011). Second, many widespread species today are undergoing massive declines e.g.

grey partridge (*Perdix perdix*; Potts, 1986; Gaston, 2010; Robinson et al., 2010), with ramifications for other species and ecosystem function/services (Gaston and Fuller, 2007). Given that the processes underlying these declines are likely to intensify in the future in many regions (Gaston and Fuller, 2007) and the potential consequences of overlooking declines in widespread species, it is imperative to identify declines in common and widespread species (Gaston and Fuller, 2007; Lindenmayer et al., 2011).

We help to meet this need by asking two questions: first, whether there have been differential population declines between widespread and restricted range species and second, whether long-term declines in widespread species are likely to be overlooked on the IUCN Red Listings.

So far, this has been difficult to achieve due to two problems: (i) we often lack sufficiently comprehensive monitoring data of populations; (ii) what data is available for conservation has been collected opportunistically where there has been a general trend for more intensive and systematic sampling in later time periods and spatial biases in recorder effort, making it difficult to disentangle changes in range size from changes in sampling effort (e.g. Donald and Fuller, 1998; Shaffer et al., 1998; Dennis and Shreeve, 2003). The solution for the first problem is to use proxies of population size such as measures of geographic range that require fewer data to construct and to examine declines over long time periods. The solution to the second problem is to account for sampling biases in data collection by using correction factors, which are discussed in greater detail in the methods section. Thus, it is important to identify species that have undergone pervasive declines and whether these range declines are likely to accord with contemporary assessments of extinction risk for widespread species. If this is the case, a different approach to assessing declines in widespread species may be needed.

We examine whether this is the case by testing: (i) whether geographically widespread species have declined at similar rates as geographically range-restricted species between pre- and post-1980 (timespan of 383 years); (ii) whether historical declines accord with current IUCN Red List categories, which will allow us to interpret short-term assessments of conservation status against long-term patterns of species decline. We do this using the bird taxon Galliformes, which is a diverse species group with a wide variety of range sizes that are likely to have had the same drivers of decline (Keane et al., 2005; Brook et al., 2008).

4.3 Methods

4.3.1 Database information

We used a historical dataset of Galliformes called GALLIFORM: Eurasian Database V.10 (Boakes et al., 2010, http://dryad.org). This database was collated from a wide range of sources with 161,025 global records from 94 species across a time span of 383 years and was geo-referenced to half a decimal degree or approximately 50 km at the equator in accuracy (thus the maximum locational error was 50 km although many records were more accurate than this). This is a comprehensive database, making it ideal for testing broad biogeographic hypotheses. The data was opportunistically collected with known biases in time and space as described by Boakes et al. (2010). Most importantly, recording effort has increased through time (see Figure 4.A1 in chapter appendix section 4.6.1). For further information on the Galliformes database and its collection, see Mace et al. (2010) and Boakes et al. (2010).

4.3.2 Assessing declines: using changes in geographic range as a proxy for changes in population size

We used changes in geographic range size as a proxy measure for population changes (Telfer et al., 2002; Kotze and O'Hara, 2003; Joseph and Possingham, 2008). Although the processes and mechanisms of range contraction can vary (Channell and Lomolino, 2001), diminishing ranges have been shown to be correlated with declines in population size (Donald and Fuller, 1998; Shaffer et al., 1998; van Swaay et al., 2008), which may lead to an increased risk of global extirpation through the successive loss of local populations (Caughley and Gunn, 1996; Purvis et al., 2000; Payne and Finnegan, 2007). Given that it is difficult to detect important aspects of range declines such as range fragmentation in widespread species when using 'Extent of Occurrence' (EOO), we chose to use a grid-based 'Area of Occupancy' (AOO) approach (further details on range measures are available in chapter appendix section 4.6.2).

4.3.3 Correction factor techniques

Correction factor techniques are known as good ways of assessing range change for opportunistically collected data with known sampling biases (Botts et al., 2012). The correction factor technique developed by Telfer et al. (2002) has already been used with this dataset for different aims and objectives (Chapter 3; Mace et al., 2010) but may under-perform in some instances, particularly against technique developed by Rich and

Woodruff (Botts et al., 2012). Specifically, while the Telfer technique is appropriate for assessing range changes over small geographic scales, such as the Himalaya (see Chapter 3), it may overestimate the largest declines over the biggest geographic scales. Thus, although the Rich and Woodruff technique gives exactly the same results as the Telfer technique in Chapter 3, in this chapter we chose to use the Rich and Woodruff technique (Rich and Woodruff, 1992) to correct for sampling biases over a global scale.

The Rich and Woodruff method can be used to calculate measures of absolute percentage change in range sizes, which is the same measure of range decline used by the IUCN (IUCN, 2012). Essentially, it assumes the later time period is more intense in terms of survey effort and seeks to correct the early time period range estimate (Rich and Woodruff, 1992). A grid-based network of half degree grid cells sampled in both time periods was used as a starting point. This also allowed us to avoid using measures of range size such as EOO, which might not fully reflect range declines. Then, equation 4.1 was used to calculate R, the new corrected historical range size estimate where N is total number of grid cells sampled (N = 2832; a subset of cells sampled in both time periods), A is original measure of historical range size and C is a correction factor based on total number of current cells/ total number of historical cells.

$$R = CA + \left\{(1 - C) + \left(\frac{A^2}{N}\right)\right\} \tag{4.1}$$

We excluded species with fewer than five cells in the early time period because they had a far greater capacity to expand than decline (after Telfer et al., 2002). In order to test the validity of this assumption, we also present the results without this omission (see chapter appendix sections 4.6.3 and 4.6.4). Finally, we repeated our analysis using the Telfer technique to ensure the results from Mace et al. (2010) were reproducible (see chapter appendix section 4.6.5 for further details).

4.3.4 Time bins

Galliformes sighting records were split into pre- and post-1980 time bins (see Figure 4.A1 in chapter appendix section 4.6.1). These bins were chosen because the published literature states that the year 1980 marks the onset of a big rise in the impact of anthropogenic threats on biodiversity (Millennium Ecosystem Assessment, 2005) and makes our analysis directly comparable to that of Mace et al. (2010). A Wilcoxon signed ranks test with continuity correction showed that the later time period had significantly more records than the earlier time bin justifying our use of the Rich and Woodruff correction technique (W = 4832.5,

$p<0.001$, $Median_{pre\text{-}1980}$ = 265 records $year^{-1}$ bin^{-1}, $Median_{post\text{-}1980}$ = 1371 records $year^{-1}$ bin^{-1}).

4.3.5 Threshold definitions

Defining species as 'range-restricted' or 'widespread' is difficult and ultimately arbitrary. Our chosen cut-off point could influence whether either group appeared to have undergone differential range changes. Thus, threshold definitions require the consideration of two factors: 1) our choice of areal threshold and 2) the time period to which we apply the threshold. Mace et al. (2010) use the median historical (pre-1980) range size, as measured through a species' Extent of Occurrence (EOO) to partition. We conducted our analysis using this median threshold definition (1,852,717 km^2), but also recognise that there are other metrics could be implemented, such as the BirdLife 50,000 km^2 threshold (Stattersfield, A.J., Crosby, M.J., Long, A.J. and Wege, 1998), which has been used to identify restricted range species across multiple studies and objectives (e.g. Abbitt et al., 2000; Olson et al., 2001; Pimm et al., 2006). We conducted our analysis using various threshold and range metrics (details of these additional metrics and why they were examined are in the chapter appendix section 4.6.2). We only present results created by different thresholds where the results were dissimilar, otherwise we only present results created by the median threshold that are directly comparable to the earlier study by Mace et al. (2010).

4.3.6 Statistics

We used two different methods to examine whether there were differences in range declines between widespread and restricted range species. First, we grouped species into two categories (either 1 = 'widespread' or 2 = 'restricted' range) and tested for differences using Wilcoxon signed ranks tests (Method 1). Second, we looked at differences in range size using continuous measures, hoping to eliminate some of the difficulties of categorising species as widespread or restricted using thresholds (Method 2). To do this, we tested whether the best fit line slope between the pre- and post-1980 range measures differed statistically from a 1:1 line with a slope of 1 and intercept of 0. We did this by first creating a 'null' model of the form $y = 0 + 1x$ and then comparing this against the 'full' model using the 'F' statistic to test for the improvement of the full model over the null, reducing the residual sum of squares by the sum of squares explained by the full model. Thus, if the best fit line differed statistically from the 1:1 line (i.e. we rejected the null hypothesis that $\alpha = \beta = 0$), it suggested that on average, range sizes had either expanded or

declined between the two time periods. Where the best fit line differed from the 1:1 line, we investigated whether differences were due to a difference in slopes or intercepts, as differences in intercepts only are unlikely to indicate differential range changes between widespread and restricted species.

For species that underwent declines, we tested whether the proportion of historically widespread species currently listed as threatened ('Vulnerable', 'Endangered' or 'Critically Endangered') by the Red List was different to those that were historically restricted. Thus, if a greater proportion of species that both have undergone declines and were historically restricted range are listed as threatened, it could indicate that different approaches to assessing declines in widespread species is needed. This was undertaken using a 2-sample, 2-tailed equality of proportions test with continuity correction, following a power analysis that showed we had sufficient replicates to detect a large effect size. All statistics were carried out using R version 2.15.1 (R Development Core Team, 2012).

4.4 Results

4.4.1 Have geographically widespread species declined at similar rates as geographically range restricted species?

Our main result was that both methods found no difference in Galliform range declines between widespread (ws) and restricted range (rr) species when the Rich and Woodruff correction factor technique was used. Method 1 gave no statistical difference between the two groups (Median threshold: N_{ws} = 48, N_{rr} = 48, $median_{ws}$ = -14.00, $median_{rr}$ = -2.90, W = 1320, p = 0.22; Bird Life threshold: N_{ws} = 92, N_{rr} = 4, $median_{ws}$ = -3.62, $median_{rr}$ = -15.48, W = 163, p = 0.71). Any differences in Method 2 are attributable to differing intercepts and not to differences in range sizes (as shown by different slopes) between the two groups over the two time periods (see Tables 4.1 and 4.2). In all cases, the best fit line is below the 1:1 line in range restricted species suggesting declines, but above the 1:1 line in widespread species suggesting expansions (see Figure 4.1). Nevertheless, on average there is no statistically significant difference in range changes.

We also found that consistent with the results in Botts et al. (2012), the Telfer technique overestimates the biggest declines relative to the Rich and Woodruff technique (Figure 4.1); this justifies our decision to use the Rich and Woodruff method. When species with less than five cells in the early time bin are included in the analysis, we find results consistent with the idea that the excluded species have a far greater capacity to expand than decline, justifying their exclusion (see Tables 4.A1, 4.A2, 4.A3, 4.A4 and 4.A5 in chapter appendix sections 4.6.3

and 4.6.4).

Technique	Residual DF	RSS	DF	SS	F	p-value
RW CF	96	35.07	2	4.10	6.20	<0.01
	94	30.98				
Raw	96	31.55	2	0.60	0.90	0.40
	94	30.95				

Table 4.1: ANOVA table showing the results of an F test between the null and full linear models. This shows that either the slope or the intercept of the best fit line is statistically different from a 1:1 line for the Rich and Woodruff corrected (RW CF) range sizes i.e. we reject the null hypothesis α=β=0. However, there is no statistically significant difference for the uncorrected range estimates (raw).

Technique	Parameter	Estimate	SE	t	p-value	F	Adj-r^2
RW CF	Intercept	-0.45	0.20	-2.27	0.03	$1.90_{(1,94)}$	0.01
	log(pre-1980 cells corrected)	0.07	0.05	1.38	0.17		
Raw	Intercept	-0.26	0.19	-1.35	0.18	$1.62_{(1,94)}$	0.00
	log(pre-1980 cells)	0.07	0.05	1.27	0.21		

Table 4.2: Summary table of full linear models for corrected and uncorrected range estimates. This table shows there was no significant statistical difference between the slope of the best fit line of the full models shown here) and a slope of 1. We can thus infer that the results from Table 4.1 are due to differences in intercepts, rather than slopes. We present both the range-corrected results using the Rich and Woodruff correction factor (RW CF) and the uncorrected (raw) range estimates.

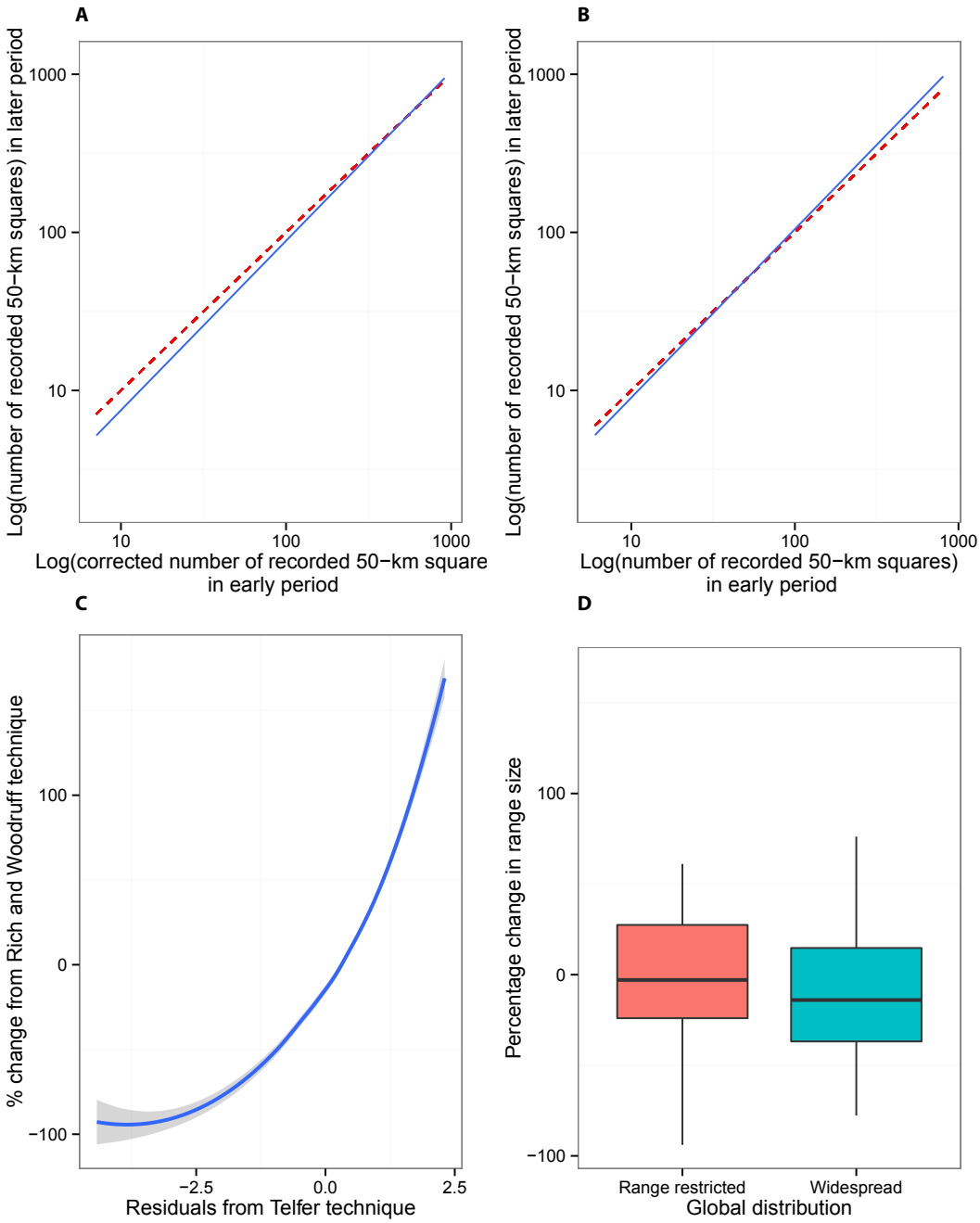


Figure 4.1: Graphs showing the results from Methods 1 and 2 and a comparison of the Telfer vs. Rich and Woodruff correction factors. The top two panels show linear regressions of range size proportions in the early vs. late period for: A) Rich and Woodruff corrected range size (number of recorded 50-km squares); B) uncorrected range size (number of recorded 50-km squares). The dashed red lines represent a 1:1 relationship between early and late range sizes and the blue lines represent the best fit line. C) shows a scatter plot with a fitted loess curve showing the relationship between the Telfer technique and Rich and Woodruff technique. This graph illustrates that consistent with the results in Botts et al., the Telfer technique overestimates the biggest declines relative to the Rich and Woodruff technique. D) shows a box and whisker plot showing the percentage range change for Galliform species with historically widespread species coloured blue and restricted species coloured red (Median threshold: N_{ws} = 48, N_{rr} = 48, $median_{ws}$ = -14.00, $median_{rr}$ = -2.90, W = 1320, p = 0.22). For all graphs N = 96 species.

4.4.2 Have historical declines been reflected in current Red List categories?

A 2-sample, 2-tailed equality of proportions test with continuity correction found that species that have declined (N = 53) were equally likely to be currently listed as threatened ('Vulnerable', 'Endangered' or 'Critically Endangered') by the IUCN Red List regardless of their historical range size (BirdLife's 50,000 km^2 threshold; $proportion_{restricted\ range}$ = 0.26 (6/23), $proportion_{widespread}$ = 0.05 (1/20), χ^2 = 2.11, df = 1, p = 0.15). However, when the median threshold was used to partition historical range size, we found that there was a reduced likelihood for historically widespread species that had declined to be listed as threatened by the IUCN Red List in comparison to historically restricted species ($proportion_{restricted\ range}$ = 0.36 (9/25), $proportion_{widespread}$ = 0.00 (0/28), χ^2 = 9.72, df = 1, p = 0<0.01).

4.5 Discussion

Widespread species are important but it is difficult to accurately assess populations change in comparison to more narrowly distributed species. Identifying whether widespread species have undergone larger historical range declines than restricted range species is important because it could have large ramifications on the way we assess extinction risk in widespread species. Similarly, if those historical declines do not accord with the current IUCN Red List categories, we might have a falsely optimistic view for widespread species listed as non-threatened ('Least Concern' and 'Near Threatened'). We found that across a group of 94 Galliform species widespread species declined at the same rate as restricted-range species (as measured across two metrics) and that accordance of these historical declines with contemporary Red List categories is not contingent on the size of a species' historical range.

In contrast to Mace et al. (2010) who conducted an analysis with different aims and objectives on the same dataset using the Telfer et al. (2002) correction factor technique, we found no difference in range changes between widespread and restricted range species. This discrepancy between range change techniques is in line with the results of Botts et al. (2012), who showed that the largest declines recorded by the Telfer technique were not recorded by the Rich and Woodruff technique (see Figure 4.1C). This effect is most prominent over larger geographic scales, given that there is a greater capacity to undergo larger declines. Given that Botts et al. (2012) found that the Rich and Woodruff correction factor was the best at identifying the sign and magnitude of known range changes within an artificial dataset, we can be confident that our results are robust. We can be sure that this difference in results is due to the correction

technique alone and not any difference between datasets as we also replicated the results of Mace et al. (2010) using the Telfer technique. Furthermore, we obtained the same result regardless of the way we measure range size or categorise species using different range thresholds. We also found the same result when we look at range changes as a function of range sized measured continuously, avoiding contentious threshold definitions of range size. Taken together, our results suggest that there are no consistent differences in historical range declines in widespread species vs. restricted species across 94 species in the Galliformes order.

We found that the link between Red List category and historical range declines was inconsistent depending on our choice of range size threshold. Thus, species with ranges smaller than 50,000 km^2 were no more likely to be listed as threatened than those with ranges above that threshold, but species with ranges smaller than 1,852,717 km^2 were more likely to be listed as threatened than those above that threshold. Interpreting these results is not straightforward, but it seems that for species that have undergone long-term range declines, those with historical ranges in size between 50,000 km^2 and 1,852,717 km^2 were consistently more likely to be listed as threatened than those with either smaller or larger historical range sizes. One possible interpretation of this result is that using 50,000 km^2 as a threshold for categorising range size does not reflect natural range patterns, but rather the imperatives of conservation planners. If we take our result that used the median range size threshold to be more ecologically meaningful, then it seems that declines in historically widespread species are less likely to accord with Red List categories than declines in historically restricted species. However, as the rate of decline between widespread and restricted species is the same regardless of how we measure range size, this lack of accord is unlikely to be a cause for concern for conservationists.

That we found that species declines were not related in a consistent way to contemporary Red List categories is not entirely unsurprising because: 1) the Red List is a composite index, so factors other than range declines might also have been used to construct the current Red List category; 2) the declines we measure are temporally discordant with Red List timescales with Red List status. Nevertheless, given that we found no consistent difference in historical range declines as a function of historical range size and that these declines were equally likely to be for both threatened and non-threatened species, it suggests that we do not need to use new tools with qualitatively different procedures to the Red List for assessing declines in widespread species. That is not to diminish the problem of missing declines in widespread species, but our results help to put the issue in context.

An analysis of range changes in the Himalayan region for Galliformes (see Chapter 3) found that long-term range changes were discordant with contemporary Red List categories; thus, non-threatened species were those that underwent the largest relative declines. It was hypothesised that this pattern could have been due in part to long-term declines in geographically widespread species being overlooked by conservationists. The current analysis suggests that this is unlikely to be the case; rather, it may be that other extrinsic drivers that have driven the pattern seen in the previous study.

There are some methodological caveats to our current analysis. First, we compared range size using time bins of unequal length, potentially giving us an unequal number of records between time periods. However, this is unlikely as the year 1980 is the median year for our records counts and is also the year where the number of records/year/time bin is approximately equal. Additionally it is preferable to use 1980 as opposed to the midpoint of the yearly range alone (midpoint = 1816 across 383 years), otherwise we would have an unequal number of records between time bins, giving the later time bin more records/year/bin.

The second caveat is that our measures of geographic range size could be affected by phylogenetic considerations. This means that more closely related Galliform species may be more likely to have similar range sizes. We treated different species as independent replicates in our study, but in doing so we could have introduced a source of pseudoreplication to the analysis. A possible solution to this could be to use phylogenetically independent contrasts, which have been used in other studies (e.g. Purvis et al., 2000).

The final caveat relates to our use of historical EOO to partition species by range size. We could argue that by using the early time period in this way, we might underestimate historical range size due to a lower sampling effort. However, this should be less of a problem than potentially biasing declines towards range restricted species by using range sizes from the late time period (see Figure 4.1). Looking at the size of the range from the data un-binned by time period is a potential solution, but any spatial changes in occupancy might inflate estimates of range size. For example, a species might have the same number of cells in each time period, but if those cells have changed in location it gives the impression of a range that is twice as big as its actual size.

In conclusion, we found no evidence for differential range declines between widespread and more restricted range species, regardless of whether we used a categorical threshold or a continuous measure to determine historical range size. Furthermore, the probability of historical range declines according with current IUCN Red List categories was not related to historical range size. Taken

together, our work suggests that special approaches are not needed to assess declines in widespread species in order to obtain a more comprehensive measure of their current conservation status.

4.6 Appendix

4.6.1 Details of the sighting data used and network of well-sampled cells

We present both the raw data (Figure 4.A1) and the network of well-sampled cells (Figure 4.A2) that it was used to construct for use in our correction-factor range-change techniques.

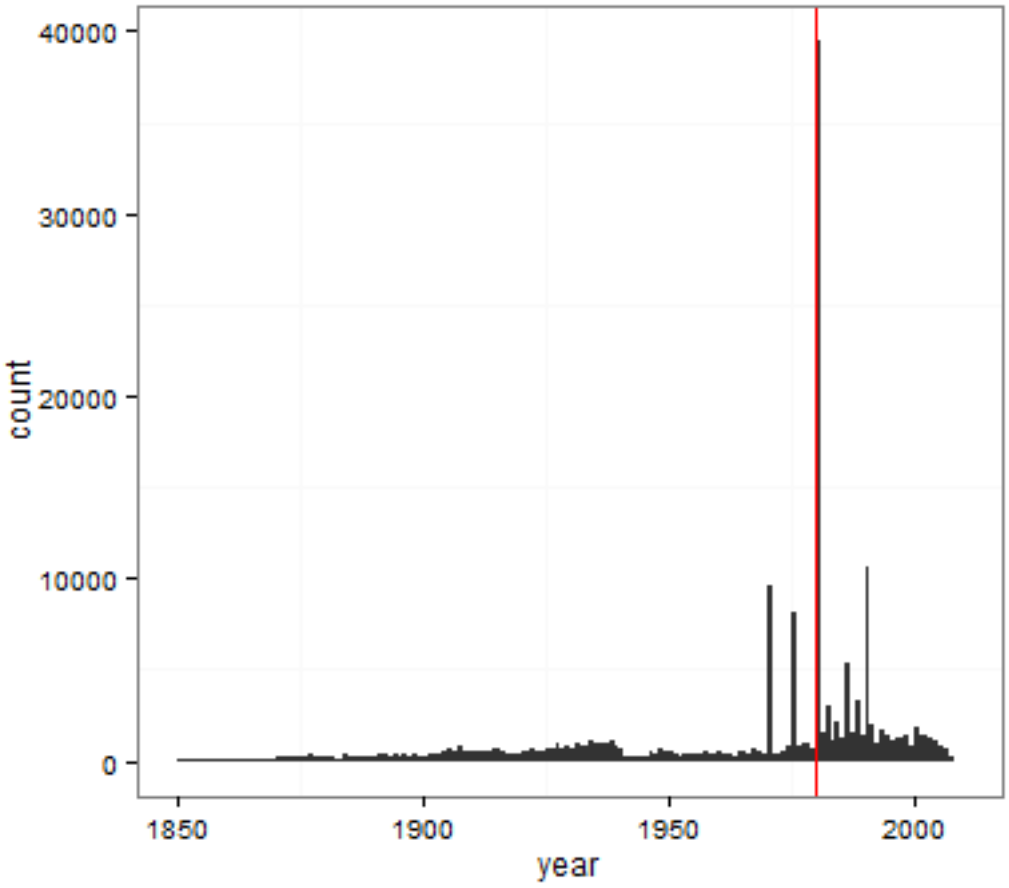


Figure 4.A1: Histogram of global record counts through time. Total number of records used = 161,023. Red line denotes median year based on number of records (1980). Length of time bins: pre-1980 = 355 years, post-1980 = 28 years. Number of records per time bin: pre-1980 = 91303, post-1980 = 69720. Median number of records/year/time bin: pre-1980= 265 records/year, post-1980 = 1371 records/year.

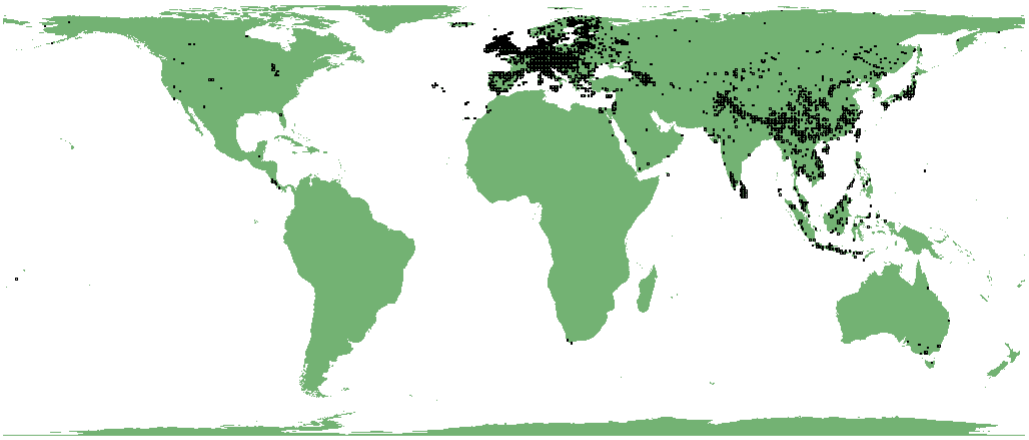


Figure 4.A2: Map showing the network of cells used in our analysis (N = 2832). Each grid cell shown was sampled from both the early and late time periods. Grid cells were approximately 50 x 50 km in dimension and were projected to a Behrmann equal area co-ordinate system.

4.6.2 Range measures

Ranges can be measured using two principle methods: Extent of Occurrence (EOO) and Area of Occupancy (AOO; Gaston and Fuller, 2009). EOO refers to the minimum convex polygon that encompasses all known normal occurrences of a particular species, whereas AOO is a subset of EOO where the species actually occurs. As AOO by definition will be a subset of EOO, reductions in AOO may affect EOO estimates very little (Gaston and Fuller, 2009). Similarly, this also means that AOO is very sensitive to grain size (Gaston and Fuller, 2009) and so for narrowly distributed species, AOO may actually be larger than EOO in some cases (see Figure 4.A3 for an example from our own data). Thus, when assessing range changes, we might a priori expect AOO to underestimate changes in narrowly distributed species and EOO to underestimate changes in widely distributed species. In order to account for both of these problems, we assessed range measures using both AOO and EOO measures. Grain size was half a decimal degree grid cell to ensure congruence with the analysis of Mace et al. (2010).

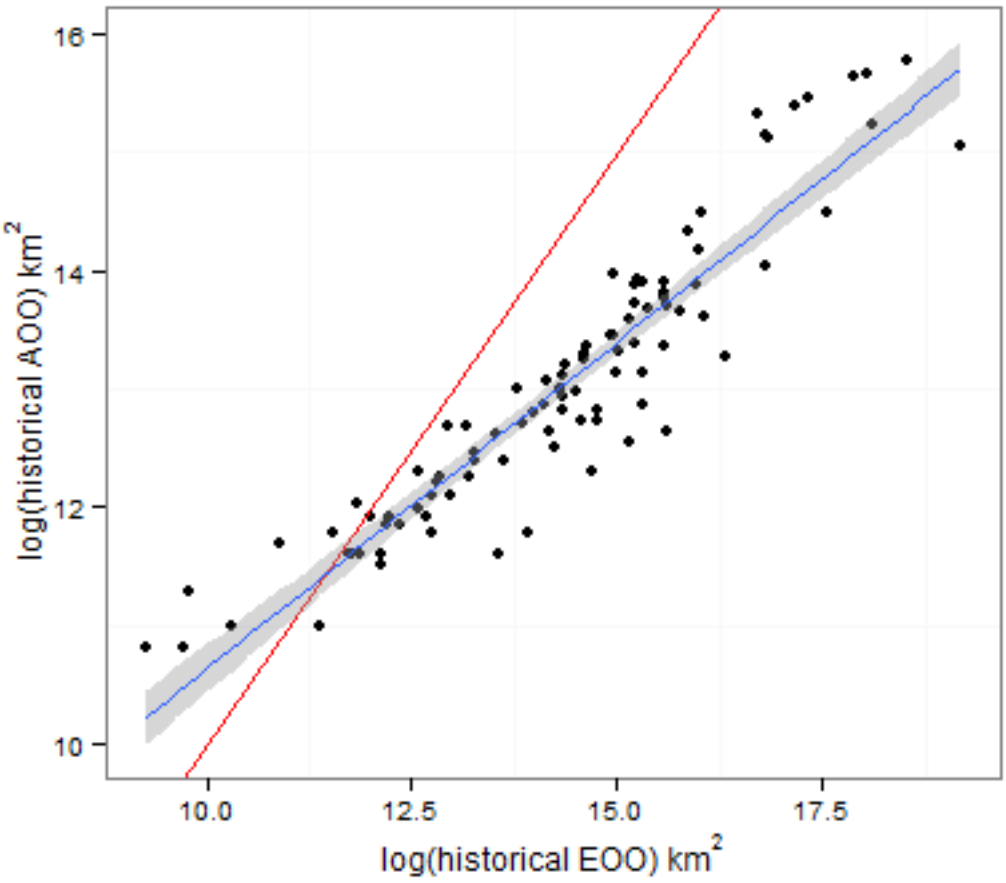


Figure 4.A3: Plot of two different range measures illustrating that for widespread species, global Extent of Occurrence (EOO) is typically much large than global Area of Occupancy (AOO); whereas the converse is true for restricted range species.

4.6.3 Additional categorical results when different threshold definitions were used and species with <5 cells in the early period were included or excluded

These results demonstrate that different threshold definitions and time bins allow us to obtain similar results. Including species with <5 cells in the early time bin did confirm the assumption that such species have a far greater capacity to expand than decline, giving the illusion that widespread species had undergone relative declines.

Threshold rule	Data examined	pre-1980 AOO_n		pre-1980 EOO		pre-1980 AOO	
		N_{ws}	N_{rr}	N_{ws}	N_{rr}	N_{ws}	N_{rr}
BirdLife	All	35	61	92	4	94	2
BirdLife	± IQR	25	25	44	4	47	2
Median	All	44	52	48	48	47	49
Median	± IQR	25	25	24	24	24	25

Table 4.A1: The number of widespread (N_{ws}) vs. the number of restricted range (N_{rr}) species as based on different threshold rules and sets of data examined (IQR = interquartile range) where species with <5 cells from early period excluded. The BirdLife threshold is 50,000 km^2 across the three range measures. The median threshold is 70,000 km^2, 1,852,717 km^2 and 390,000 km^2 across the three range measures. Key: AOO_n = area of ocupancy based on a network of well-sampled cells, EOO = extent of occurrence and AOO = area of occupancy.

Threshold rule	Data examined	pre-1980 AOO_n		pre-1980 EOO		pre-1980 AOO	
		N_{ws}	N_{rr}	N_{ws}	N_{rr}	N_{ws}	N_{rr}
BirdLife	All	61	61	106	16	108	14
BirdLife	± IQR	31	31	46	16	48	14
Median	All	61	61	61	61	61	61
Median	± IQR	31	31	31	31	31	31

Table 4.A2: The number of widespread (N_{ws}) vs. the number of restricted range (N_{rr}) species as based on different threshold rules and sets of data examined (IQR = interquartile range) where species with <5 cells from early period included. The BirdLife threshold is 50,000 km^2 across the three range measures. The median threshold is 51,250 km^2, 1,145,548 km^2 and 290,000 km^2 across the three range measures. Key: AOO_n = area of ocupancy based on a network of well-sampled cells, EOO = extent of occurrence and AOO = area of occupancy.

Data	Range measure	Correction factor	Threshold	N_{ws}	N_{rr}	$Median_{ws}$	$Median_{rr}$	W	P
All	AOO_n	Rich	BirdLife	61	61	-2.92	-15.65	1923.0	0.75
All	EOO	Rich	BirdLife	106	16	-4.82	7.76	1061.0	0.11
All	AOO	Rich	BirdLife	108	14	-7.68	15.25	998.5	0.05
Subset	AOO_n	Rich	BirdLife	31	31	-1.31	12.43	554.0	0.30
Subset	EOO	Rich	BirdLife	46	16	-4.10	7.76	475.0	0.09
Subset	AOO	Rich	BirdLife	48	14	-14.17	15.25	460.0	0.04
All	AOO_n	Rich	Median	61	61	-2.92	-15.65	1923.0	0.75
All	EOO	Rich	Median	61	61	-15.67	5.59	2416.5	0.00
All	AOO	Rich	Median	61	61	-15.90	5.59	2409.0	0.01
Subset	AOO_n	Rich	Median	31	31	-1.31	12.43	554.0	0.30
Subset	EOO	Rich	Median	45	64	-20.43	5.55	1809.5	0.01
Subset	AOO	Rich	Median	36	61	-23.90	5.55	1490.0	0.00

Table 4.A3: Wilcoxon signed rank test results. Key: AOO_n = AOO based on network of well-sampled cells, EOO = extent of occurrence, AOO = area of occupancy, subscript $_{ws}$=widespread, $_{rr}$=range restricted.

4.6.4 Additional continuous results when species with <5 cells in the early period were included in the analysis

As before, including species with <5 cells in the early period confirmed the assumption that such species had a far greater capacity to expand than decline, giving the impression that widespread species had undergone relative range declines.

Technique	Residual DF	RSS	DF	SS	F	p-value
RW CF	122	53.04	2	2.82	3.36	0.04
	120	50.22				
Raw	122	50.33	2	0.24	0.28	0.75
	120	50.1				

Table 4.A4: ANOVA table showing the results of an F test between the null and full linear models. Key: RW CF = Rich and Woodruff correction factor, raw = uncorrected range estimates.

Technique	Parameter	Estimate	SE	t	p-value	F	Adj-r^2
RW CF	Intercept	-0.07	0.15	-0.47	0.64	0.35 (1,120)	-0.01
	log(pre-1980 cells corrected)	-0.02	0.04	-0.59	0.55		
Raw	Intercept	0.10	0.14	0.75	0.45	0.46 (1,120)	0.00
	log(pre-1980 cells)	-0.03	0.04	-0.68	0.50		

Table 4.A5: Summary of full linear models for uncorrected and corrected range estimates for species with <5 cells in the early time period. These results show there was no significant statistical difference between the slope of the best fit line (full models shown here) and a slope = 1. Key: RW CF = Rich and Woodruff correction factor, raw = uncorrected range estimates.

4.6.5 Additional results relating to the Telfer method

The results shown in Table 4.A6 allows us to compare our results to the results obtained by Mace et al. (2010).

Species with <5 cells in the early period	Un-weighted regression equation	Weighted regresson equation	Number of iterations required
Excluded	y=1.07x + 0.31	y=1.07x + 0.30	4
Included	y=0.99x - 0.17	y=1.12x + 0.67	5

Table 4.A6: The number of iterations required to weight the original regression equation in order to account for heteroscedasticity using the Telfer method.

4.6.6 What is a range-restricted species? Further notes on our threshold definitions

Choice of areal threshold

We used the median values for our suite of species after Mace et al. (2010) to partition species based on range size (for details of these median values see Tables 4.A1 and 4.A2). Thus, species with range sizes above the threshold were counted as 'widespread' and those below the threshold were counted as 'restricted'. However, we also used the BirdLife definition of 50,000 km^2 (Stattersfield, A.J., Crosby, M.J., Long, A.J. and Wege, 1998) as a threshold because of its prevalence in conservation biology. Additionally, a subset of species with range sizes above and below the interquartile range for all species was tested in order to identify signals in the data more clearly (see Tables 4.A1, 4.A2 and 4.A3).

Choice of time period to which threshold definitions were applied

Our choice of threshold time bin can affect how we interpret our results because range size is used as both an explanatory and a response variable in our analysis. For example, we can measure range size as an explanatory variable based on the pre-1980, post-1980 or even the mean range size between the two time periods. Following the lead of (Mace et al., 2010), we used the pre-1980 time bin for our analysis. This is because we wanted our changes to be measured relative to the historical range size.

Chapter 5. The protected area network in the greater Himalayas needs improvement to protect Himalayan galliforms (gamebirds)

5.1 Abstract

Protected areas (PAs) are a key tool in protecting biodiversity, but optimising their placement in the landscape is challenging due to a poor knowledge of biodiversity coverage and differences in how to best prioritise biodiversity value. We assess these two issues for a highly threatened group of birds in the Greater Himalaya, which is a region of conservation concern. To determine if the existing PA network is adequate at protecting the most important areas for 24 species of Galliformes (gamebirds) found in the Himalayas, we compare the current distribution (based on an extensive data N = 2567 records collected post 1980) to that produced from Zonation prioritisation software and modelled species distributions. Specifically, we identify areas of high species richness and then we weight maps by different species-specific conservation values including Red List score, a measure of regional relative declines, endemism to the Himalaya and phylogenetic distinctiveness. We find that regardless of the type of conservation value, the North West of India, Central Bhutan and the North East/South China border are consistently important for Galliformes of conservation concern. We found statistically significantly poorer fits between the optimal solution of PA network (based on three different prioritisation methods using Red List category OR endemism to the Himalayan region OR un-weighted analysis treating each species of equal importance) and the actual Himalayan PA network (the average optimal PA network across the five different prioritisation methods, covered 56% more of the distributions of Galliform species than the current network). We suggest some refinements to the current PA network in order to maximise its effectiveness for Galliformes.

5.2 Introduction

The uneven spatial distribution of both anthropogenic threats (Cincotta et al., 2000) and biodiversity (Gaston, 2000) makes site-specific prioritisation of scarce conservation resources a necessity. This is most commonly manifested in the form of protected areas (PAs) that aim to either halt or manage such threats (Bruner et al., 2001). PAs are seen as an important contribution to

biodiversity conservation with site-based conservation reported to be appropriate for 82% of birds, mammals and amphibians (Boyd et al., 2008), although the PA coverage of species ranges can be only described as adequate and highly variable at best (Rodrigues et al., 2004). There has been a significant recent expansion in the number of PAs and the area that they cover in the last 20 years (Jenkins and Joppa, 2009). The political recognition of the importance of these areas is demonstrated in the Convention on Biological Diversity's Strategic Plan for 2011 (CBD, 2010), with the intention of extending coverage to 17% of the terrestrial surface as Aichi target 11 makes clear (see Chapter 1 Table 1.1).

Assessing how useful protected areas are for biodiversity conservation is challenging, as it requires a range of measures to be taken for a meaningful appraisal to be made in addition to percentage targets in global coverage. These measures include ensuring PAs have a suitable extent of representation, especially for areas important for biodiversity and ecosystem services; provide effective conservation through equitable management and are ecologically representative and well-connected. It is not easy to measure these characteristics so that a useful assessment of the world's 150,000 or so protected areas can be made (Woodley et al., 2012). The Biodiversity Indicators Partnership has suggested three measures are used: a) management effectiveness of protected areas; b) coverage of protected areas; c) protected area overlays with biodiversity (www.bipindicators.net/globalindicators, 2014). Existing datasets have been identified to assess these three indicators and it is acknowledged that there are challenges in developing a robust and practical way of assessing progress towards this target.

A key facet of this target is the ability to place protected areas in a way that reflects the qualities of biodiversity that we wish to conserve (clause three of Aichi target 11 commits countries to establish protected areas in locations that are of 'particular importance for biodiversity and ecosystem services'). This presents two issues: i) the parameters of biodiversity we are concerned with and ii) how well we know where biodiversity occurs.

First, different aspects of biodiversity can be valued in different ways (Nunes and van den Bergh, 2001). For example, biodiversity values that are the basis of area prioritisation programmes that have been developed over the last 20 years include endemism (Endemic Bird Areas; Stattersfield, A.J., Crosby, M.J., Long, A.J. and Wege, 1998), overall species richness (hotspots; Myers et al., 2000), representativeness (ecoregions; Olson et al., 2001) and extinction avoidance (Alliance for Zero Extinction sites; www.zeroextinction.org, 2010) to name but a few. The number and area covered by such prioritisations is so large that most of the terrestrial environment is covered by at least one (Brooks

et al., 2006), but different conservation values may be incongruent with each other (Prendergast et al., 1993; van Jaarsveld, 1998; Moritz et al., 2001; Grenyer et al., 2006). Second, our knowledge of the spatial distribution of biodiversity is often incomplete (Lomolino, 2004; Whittaker et al., 2005) and uncertainties in the spatial distribution of biodiversity may not be acknowledged in systematic spatial conservation planning (Polasky et al., 2000; Gaston and Rodrigues, 2003; Rocchini et al., 2011).

We assess these two issues for a set of birds that belong to a group that contains 25% threatened species (Keane et al., 2005) and in a region that is of increasing conservation concern (Pandit et al., 2006; Hoffmann et al., 2010), and a target for protected area expansion (Venter et al., 2014). The Greater Himalaya provides important habitat for around 24 species of the bird Order Galliformes (ENVIS, 2007), which are found throughout the entire Himalaya (ENVIS, 2007) and have a range of conservation statuses. They are important both culturally (Baral, 2009; Ramesh and McGowan, 2009) and as a source of protein (Keane et al., 2005). More broadly, the Greater Himalaya has been identified as an important conservation region for birds due its high levels of species richness and endemism (Chan et al., 2004; Brooks et al., 2006; Xu et al., 2009). Simultaneously, the Greater Himalaya has increasingly come under threat from a rapidly expanding human population, which has been manifested in an extremely high proportion of threatened terrestrial species (Hoffmann et al., 2010).

The current Himalayan protected area (PA) network has been shown to overlap with the ranges of all Himalayan Galliform species in some way (McGowan et al., 1999). Simply representing all species within the current Himalayan PA network at least once might be inadequate for Galliform conservation in three main ways: first, this approach ignores the likelihood of species occurrence and focuses on binary presence/absence data. Therefore, while every species may occur in the PA network, not all species may be represented equally in terms of optimal habitat quality and/or suitability (Rondinini et al., 2005, 2006) or represent coverage of sustainable populations, leaving important external populations vulnerable to threats (Witting and Loeschcke, 1995; Pressey et al., 2004). Second, there may be incongruence between the locations of important areas depending on our definitions of species-specific conservation values, such as species richness, endemism and threatened species (Orme et al., 2005; Ceballos and Ehrlich, 2006). These values are not always considered a scientific matter (Miller et al., 2006), reflecting the underlying values of humans, rather than nature (Vane-Wright, 2009). This can be problematic because conservation planners may have multiple objectives, ranging from preserving biodiversity patterns to protecting species at the greatest risk of extinction (Arponen, 2012).

Therefore, although Galliform species are represented within the Himalayan PA network, we may not be adequately capturing different types of species value (Terribile et al., 2009). Third, while every Galliform species may be represented in a protected area within the Himalaya, these protected areas have recently come under criticism for failing to adequately enforce legislation, especially regarding habitat conversion (Clark et al., 2013). This may mean protection from threats is in name only.

Therefore there is a need to assess whether the current PA network represents the most important areas for conservation within the Himalaya for Galliformes based on a variety of types of conservation value and to assess whether the current protected network adequately represents these species of conservation concern. To achieve this aim, the present study combines niche modelling software with reserve selection algorithm software. First, we map Galliform distributions in the Himalayas by creating environmental niche models for 23 of the 24 species that are found there. Second, we identify the areas of greatest conservation importance to these species based on a range of different criteria before we assess the representation of these important areas within the current protected area (PA) network.

5.3 Methods

5.3.1 Species distribution modelling procedure

Maxent is a machine-learning method that is used to produce niche models (version 3.3.3k; Phillips et al., 2006). Niche modelling predicts a species' geographic distribution as a function of occurrence records and environmental data layers (Guisan and Zimmermann, 2000; Rushton et al., 2004; Guisan and Thuiller, 2005) and can be particularly appropriate where sampling biases in occurrence records make it difficult to distinguish between a lack of survey effort vs. a lack of species occupancy (Polasky et al., 2000; Funk and Richardson, 2002; Lombard et al., 2003). Maxent has been found to perform well against other distribution models (Elith and Graham, 2009) and produces outputs that have particularly high accuracy.

We created Maxent environmental niche models for 23 of the 24 species (we were not able to create a niche model for the Himalayan quail) of Galliformes within the Greater Himalayas that had sufficient point locality data (i.e. more than 10 records since 1980; Table 5.A1 in chapter appendix section 5.6.1). These models were created using point locality data from the GALLIFORM: Eurasian Database V.10 (http://dryad.org; Boakes et al., 2010) for 23 species of Galliformes that occur in the Greater Himalaya, which was accurate to

approximately 1 km and collected from a wide range of sources including museum specimens, ringing records, biological atlas data and trip reports. We omitted records without locality information or that had a locational error greater than 1 km. For further details on the number of locality records available for each species see Table 5.A1 in chapter appendix section 5.6.1. We only used post-1980 data to ensure temporal concordance between measures of land cover and sighting records (Guisan and Zimmermann, 2000; Boitani et al., 2011).

We used a range of variables chosen through expert consultation (via members of the IUCN Galliform Specialist Group – for details see the chapter appendix section 5.6.1) as simple surrogates to cover the main potential determinants of bioclimatic variation. We used a subset of four bioclimatic variables (mean annual temperature, temperature seasonality, mean annual precipitation excluding snow and variation in precipitation) that were downloaded from www.worldclim.org/bioclim (2014), three topographic variables (elevation, slope and aspect) that were downloaded from the 90 m Shuttle Radar Topography Mission (SRTM) at 30 arc seconds (Jarvis et al., 2008). As a non-categorical summary of land cover we used Normalised Difference Vegetation Index (NDVI) variables from the SPOT Vegetation sensor. Nine years of data (1999-2007) were downloaded from www.vito.be and were combined to create an average monthly NDVI raster. Model variables were standardised to the same spatial scale (1 km^2) and processed using ArcMap version 10.2 and projected to a South Asia Albers equal area projection. Variables selected for inclusion in the final models were those that contributed $>$3% to the maximal model to avoid over-fitting the models while maximising their predictive power. No attempts were made to omit collinear variables as machine learning methods have been shown to still perform well with such variables, especially when the study goal is predictive accuracy (Elith et al., 2011).

Our overall study site was delimited based on WWF Ecoregions (for full details see Table 5.A2 in the chapter appendix section 5.6.1; Olson et al., 2001). We incorporated ecoregions as a categorical variable in the analysis to prevent extrapolation beyond the focal regions in which species occurred. Regularisation values were chosen based on AICc (Warren and Seifert, 2011). Our choice of feature function was determined by the smoothness of the response curves and also by the number of sample points (Table 5.A3 in chapter appendix section 5.6.1; Phillips and Dudík, 2008). We used locations from which there were other Galliformes (N = 2567 records) to generate a targeted set of pseudo absences. Thus our pseudo absences were chosen from sites with the same sampling bias as the presences for a suite of species that

were observed with similar sampling techniques. This 'target group' approach (Phillips and Dudík, 2008) reduces the potential for species distribution model outputs to be affected by sampling biases in study species records in both time and space (see Boakes et al., 2010).

The ability of each model to discriminate between occupied and unoccupied areas was estimated from the area under the curve (AUC) of the receiver operating characteristics (Phillips et al., 2006). Ten cross validations were undertaken to generate folds of randomly-selected presence data, allowing us to run the model ten times, exclude each fold in turn and using the fold to validate the data (Phillips and Dudík, 2008). These model outputs were then clipped to the overall study region for use in Zonation (see chapter appendix section 5.6.2).

5.3.2 Identifying important conservation areas as measured by different conservation values using Zonation and accounting for uncertainty

Zonation produces a complementarity-based hierarchical prioritisation of the landscape based on the biological value of sites (Moilanen et al., 2005; Moilanen, 2007). This complementarity-based algorithm estimates the optimal set of areas that are together as valuable as possible, while accounting for differences, similarities and connectivity between different sites. The Zonation hierarchy is generated by the iterative removal of cells, whose loss causes the smallest decrease in conservation value in the remaining network. This algorithm can be tailored through priorities and connectivity responses defined by the user and assigned to biodiversity features in the analysis (Arponen et al., 2005). In addition to a nested graduation of conservation value throughout the landscape, an associated set of curves that describes how well each species does at any given level of cell removal is produced. The graduated zones within a landscape correspond to different degrees of conservation value and may be used as a guide to determine the level of protection needed. This differs from previous target-based planning or maximum-coverage approaches as it is a hierarchy of nested solutions, rather than a single optimal solution (Moilanen, 2007). Zonation has been used before in the identification of important areas for Italian butterflies (Girardello et al., 2009) and for fish in New Zealand (Moilanen et al., 2008) and has been found to give comparable results to other systematic conservation planning software such as Marxan, while retaining a focus on the connectivity of sites (Delavenne et al., 2012).

Our basic settings (prioritising areas with the highest species richness) were weighted by four measures of species-specific conservation value (Table 5.A5 in chapter appendix section 5.6.3; for a similar approach see Girardello et al.,

2009): i) IUCN Red List; ii) regional range change scores; iii) endemism to Himalaya; iv) phylogenetic distinctiveness. Red List scores represent a composite measure of global extinction risk, range changes scores represents a simple measure of regional relative range declines, endemism scores represent a measure of how important the Himalaya is for each species in terms of geographic range and phylogenetic scores represent a measure of evolutionary distinctiveness (full details of relative weights are given in the chapter appendix section 5.6.3). In absence of weighting data (e.g. regional range change scores), we left scores un-weighted.

Uncertainty in data inputs were accounted for by weighting species by the confidence we had in the Maxent outputs based on expert opinion and smoothness of AUCs (area under the curve) of the receiver operating characteristics (see the chapter appendix section 5.6.3; Phillips et al., 2006). Where we were unable to generate a full niche model for species such as the Himalayan quail, we used the Species of Special Interest (SSI) feature to include point locality data in the place of a species distribution map (Moilanen, 2012). We also used the standard deviation outputs from the Maxent model cross validations as a measure of uncertainty in a distribution discounting analysis. Thus for each species, we subtracted the model standard deviation from its respective niche model. This had the effect of retaining only the Maxent outputs with the highest certainty in the final Zonation solution (for details see Moilanen et al., 2006).

We compared the spatial distribution of different Zonation solutions and examined representation curves for each measure of conservation value. The top 18.1% of the study area for each solution was extracted and compared by way of the Kappa statistic, a measure agreement between categorical outputs using Map Comparison Kit 3.2.3 (Visser and de Nus, 2006) and a spearman rank correlation of the continuous outputs using ENMTools (Warren et al., 2010, 2008). The 18.1% figure was chosen to represent the most important areas of the study area based on the current proportion of the Himalayas taken up by protected areas, which also is close to the Aichi Target 11 figure of 17%.

5.3.3 Assessing the representation of important conservation areas within the protected area network

Protected Area shapefiles were downloaded (see Figure 5.1 for a map of the current Himalayan PA network; www.protectedplanet.net, 2014) and used in the same Zonation analyses as before but with the additional constraint of the top fraction of the landscape being forced through the existing PA network (for examples see Cabeza and Moilanen, 2006; Kremen et al., 2008; Leathwick et al.,

2008). Thus, if important conservation areas are spatially congruent with the PA network, the top fraction of the landscape should be in the same place. We compared representation curves at the level of cell removal (proportion of landscape lost) that corresponded to the geographic area of our existing protected area network. This was 118,543 km^2 out of a possible 654,772 km^2, corresponding to 18.1% of all land in our study site. This comparison was made across all species and shown both as an average value for all species and the worst off species only.

Finally, we compared the top 18.1% of outputs from differently weighted solutions to see how much they overlapped with the current PA network. If important areas of Galliform biodiversity are represented significantly less well in the current protected area network in comparison to the optimal solution, it might suggest that protected areas are situated in the wrong place.

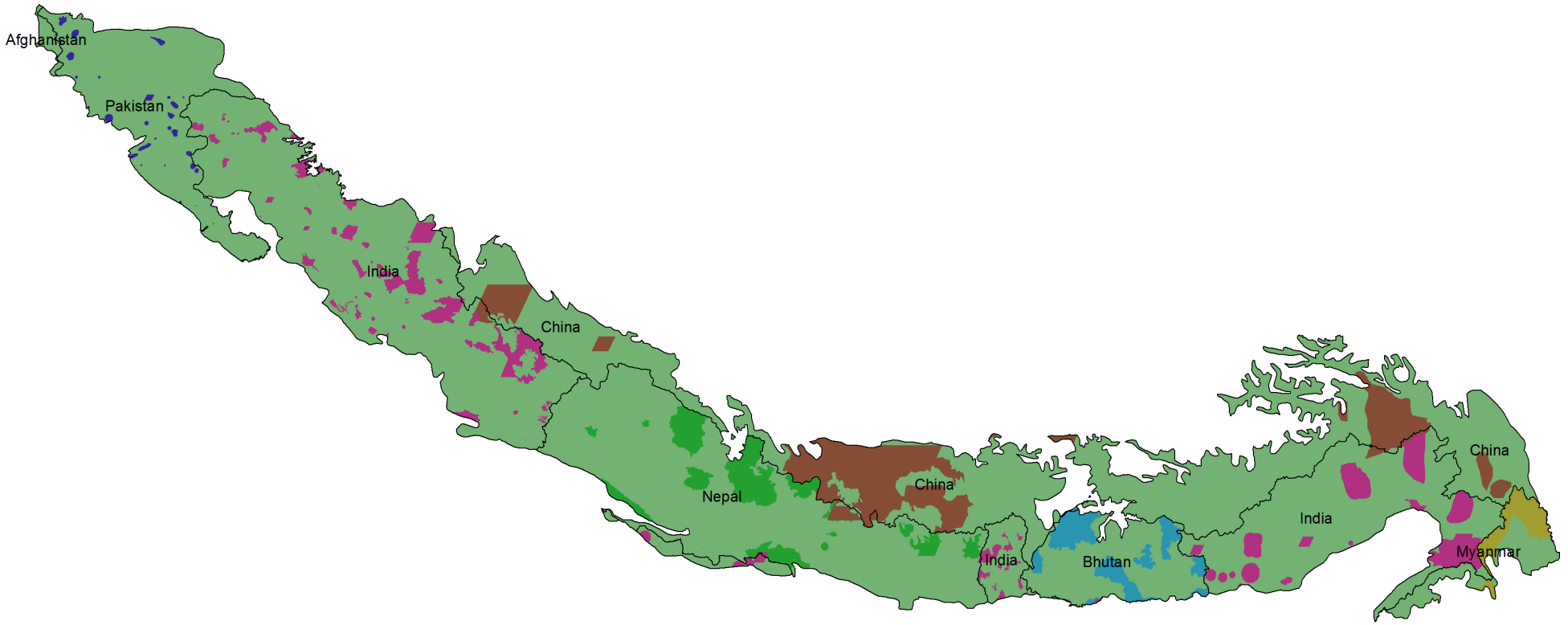


Figure 5.1: Map of current Himalayan PA network. Note: different PAs are coloured by the country that they are located within. 98% of Himalayan PAs are trans-national or have borders that lie alongside country boundaries.

5.4 Results

5.4.1 Mapping Galliform distributions within the Himalaya

The modelled distributions indicated that Galliformes were distributed across the entire study area (654,772 km^2) with particular concentrations in North West India, Central Bhutan and North East India and along the southern border of China (Figure 5.2). Individual maps for each species are shown in the chapter appendix section 5.6.2 (Figures 5.A1-5.A3).

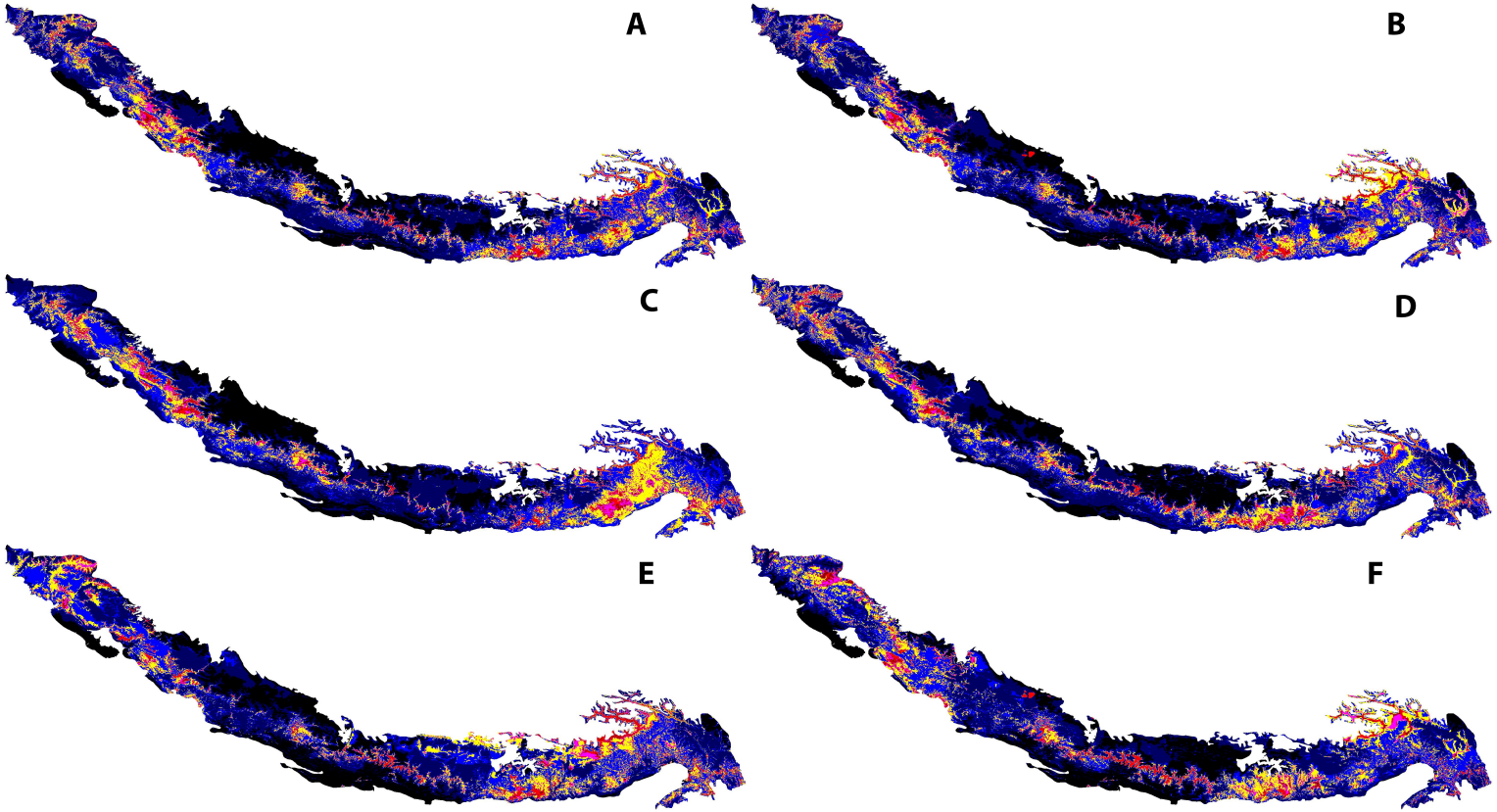


Figure 5.2: Zonation output maps under various scenarios (A = basic, B = distribution discounting, C = weighted by Red List, D = weighted by endemism, E = weighted by range change, F = weighted by phylogenetic distinctiveness) for species of Galliformes occurring in the Himalayan region. The warmest colours indicate the most important conservation areas and the coolest colours indicate the least important conservation areas: black = 0-20% (least important areas), dark blue = 20-50%, blue = 50-75%, yellow = 75-90%, pink = 90-95%, dark red = 95-98% and red = 98-100% (most important areas). The top fraction (18.1%) of each solution corresponds to an area of approximately 118,513 km^2 out of a total of 654,772 km^2, which is equal to the current area taken up by PAs. Note: weighted solutions presented here were weighted after accounting for data uncertainty using distribution discounting.

5.4.2 Identifying important conservation areas and accounting for uncertainty

We show different zonation solutions that account for data uncertainty and different types of conservation value (Figure 5.2). Our results highlight how the areas of high species richness reported above are consistently important for Galliformes of conservation concern, regardless of which of the four types of conservation value were examined.

Examination of details did highlight some statistically significant differences between types of conservation value (see Table 5.1 and Figures 5.A5-5.A6 in chapter appendix section 5.6.4). On its own, weighting by Red List places greater importance towards areas in Eastern India/Myanmar whereas weighting by endemism places greater importance on areas in Bhutan/Eastern India. Areas weighted by phylogenetic distinctiveness do not overlap with areas weighted by relative range declines in the North West of the Himalaya (Figure 5.3). Similarly, areas weighted by endemism do not overlap with areas weighted by Red List in the South East of the Himalaya (Figure 5.3). The area of overlap between different zonation solutions is 29,688 km^2, which is 4.5% of the total study site area and 25.2% of the area taken up by the top fraction (i.e. top 18.1%) of any one solution. Of this area of overlap between every different zonation solution, 23.6 % (6993 km^2) falls within the current PA network, which corresponds to 1.1% of the total study site area.

Overlap between the basic solution and the solution distribution discounting based upon uncertainty in predicted range is very high (Table 5.1), suggesting that accounting for uncertainty in Maxent outputs does not change the spatial location of the Zonation output greatly.

Comparison	Categorical				Continuous		
	Kappa	$K_{location}$	$K_{histogram}$	Fraction agreement	r^2	Intercept	Slope
BA x DD	0.79	0.79	1	0.94	1.00	0.00	1.00
DD x EN	0.53	0.53	1	0.86	0.84	-0.34	0.67
DD x RL	0.49	0.49	1	0.85	0.99	-0.01	0.99
DD x PD	0.55	0.55	1	0.87	0.99	-0.01	0.99
DD x RRC	0.56	0.56	1	0.87	0.99	-0.01	0.99
EN x RL	0.61	0.61	1	0.88	0.99	0.26	1.26
EN x PD	0.33	0.33	1	0.80	0.97	0.04	1.04
EN x RRC	0.44	0.44	1	0.80	1.00	0.12	1.12
RL x PD	0.24	0.24	1	0.77	0.98	-0.01	0.98
RL x RRC	0.34	0.34	1	0.80	0.99	-0.01	0.99
PD x RRC	0.39	0.39	1	0.82	0.98	-0.01	0.98

Table 5.1: Spatial comparison of different Zonation solutions based on both continuous output (spearman rank coefficient, intercept and slope) and categorical output of top 18.1% of the solution (Kappa, $K_{location}$, $K_{histogram}$ and fraction agreement). Zonation solution codes: BA = basic, DD = distribution discounting, EN = endemism, RL = Red List, PD = phylogenetic distinctiveness and RRC = relative range change. Values of Kappa (after Landis and Koch, 1977): < 0 = no agreement, 0–0.20 = slight, 0.21–0.40 = fair, 0.41–0.60 = moderate, 0.61–0.80 = substantial, and 0.81–1 = almost perfect agreement.

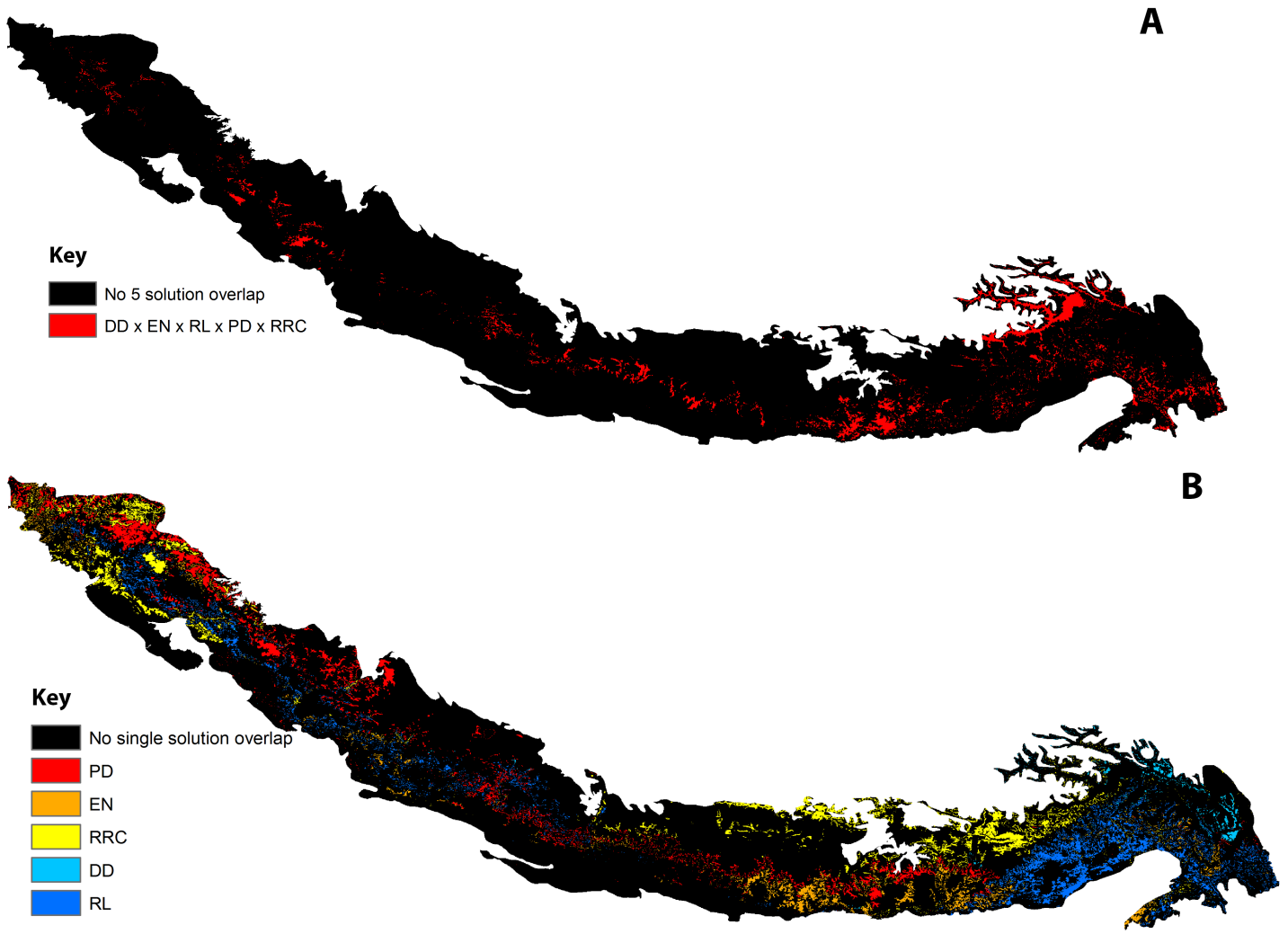


Figure 5.3: The level of overlap between different Zonation solutions (A = overlap for all solutions, B = single solutions only). For a more detailed breakdown of the areas involved see Tables 5.A5-5.A6 in the chapter appendix. Most of the areas are coloured black which indicates a lack of overlap between solutions for each respective scenario whilst the different colours for each scenario show the extent of overlap. The area of overlap between all solutions corresponds to an area of approximately 29,688 km^2 and 23.5% of this area overlaps with the current PA network. The Zonation solution codes: DD = distribution discounting, EN = endemism, RL = Red List, PD = phylogenetic diversity and RRC = relative range change.

5.4.3 Assessing the representation of important areas for Galliform conservation within the protected area network

The results presented in Table 5.2 and Figure 5.4 indicate that there is statistical evidence to suggest that the current PA network provides a poorer fit than the optimal solution derived via three out of five different conservation prioritisation methods. Both the un-weighted distribution discounted solution and the solutions weighted by endemism and Red List respectively, show the unconstrained solution to retain almost twice the proportion of landscapes in comparison to the constrained solution (Table 5.2 and Figure 5.4). Thus, the distributions of Galliform species at the greatest risk of global extinction and those with the narrowest endemism to the Himalaya are less well covered by the current PA network than is optimal.

	Average				Minimum			
Solution	**Unconstrained**	**Constrained**	χ^2	**p-value**	**Unconstrained**	**Constrained**	χ^2	**p-value**
DD	0.30	0.19	2.74	0.05	0.10	0.09	0.00	0.50
EN	0.30	0.18	3.06	0.04	0.09	0.08	0.00	0.50
RL	0.30	0.18	2.92	0.04	0.09	0.08	0.00	0.50
PD	0.26	0.18	1.50	0.11	0.15	0.10	0.70	0.20
RRC	0.28	0.19	2.07	0.08	0.10	0.09	0.00	0.50

Table 5.2: A comparison of the proportion of distributions remaining between the unconstrained and constrained solutions at the level corresponding to the protected area network (18.1%). A 1-tailed equality of proportions test revealed that the unconstrained solutions averaged across all species contained a significantly greater proportion of all species distributions than the constrained solutions for DD ('distribution discounting'), EN ('endemic species') and RL ('Red List') species (proportions are given to 2 d.p., hence small differences between χ^2 values between average solutions 'EN' and 'RL'). However, for the worst-off (denoted as 'minimum' in the table) species only, there were no significant differences between constrained and unconstrained solutions across all types of conservation value. Note: the overall size of the output areas between the unconstrained and constrained solutions remain constant and only the proportion of species distributions represented by each solution changes. The mean increase in distributions covered in the unconstrained vs. the constrained solution across the five methods was 56%.

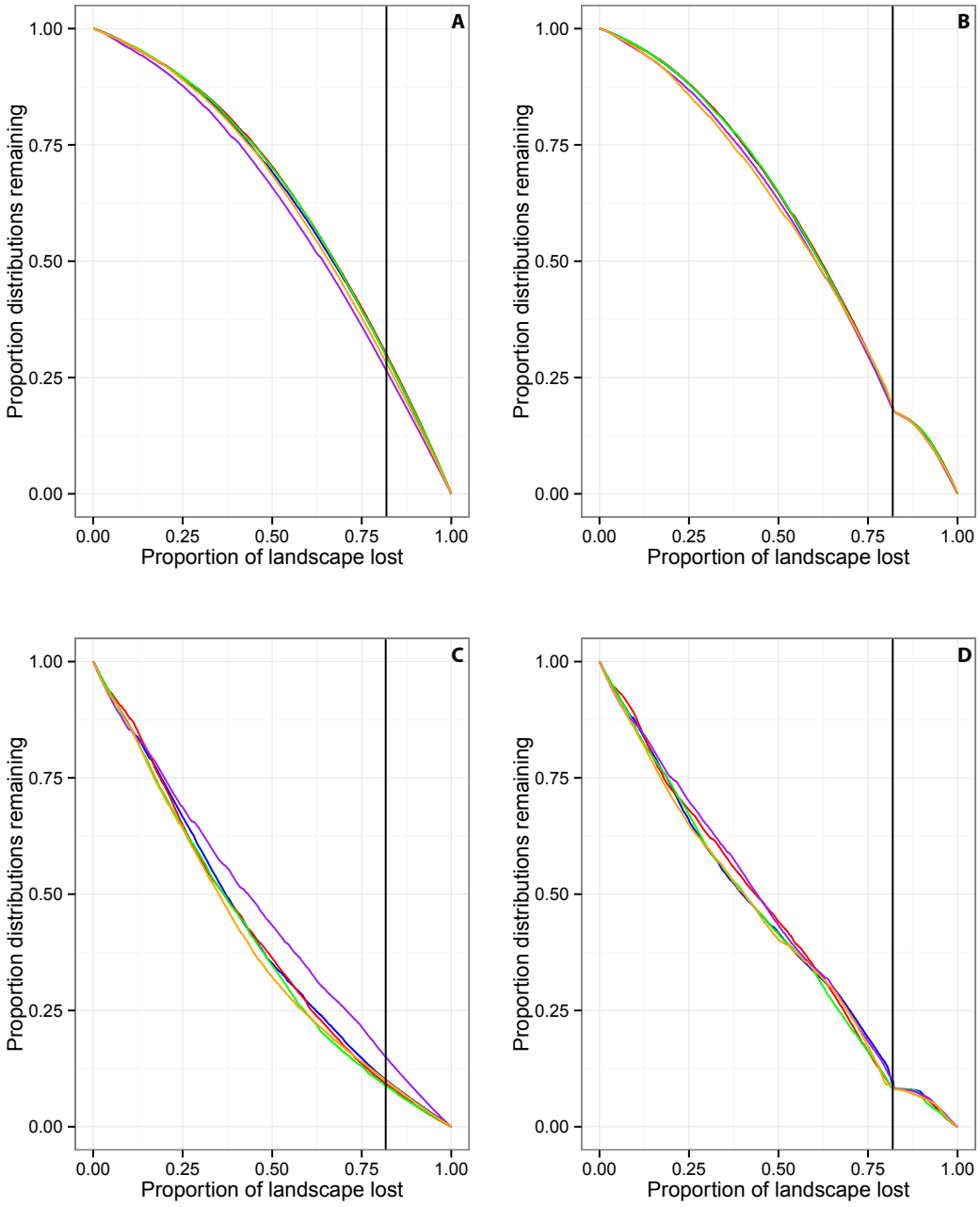


Figure 5.4: The top two panels show the average Zonation response curves across all species for both A) unconstrained and B) constrained solutions. The bottom two panels show the minimum response curves for the worst off species for both C) unconstrained and D) constrained solutions. Response curve colours correspond to the following Zonation solutions: blue = distribution discounting, red = endemism, green = Red List, purple = phylogenetic distinctiveness and orange = range change. The vertical lines represent 1-proportion of the Himalayan landscape taken up by PAs (i.e. 81.9% of the landscape lost). Comparisons between the proportion of species distributions where the response curves intersect the vertical line (A vs. B and C vs. D) indicate the efficacy of the current PA network in capturing Galliformes distributions. i.e. if PAs were perfectly placed, the proportions of distributions remaining would be equal.

5.5 Discussion

Protected areas (PAs) have a vital role to play in achieving global biodiversity targets (Rodrigues et al., 2004; Secretariat of the Convention on Biological Diversity, 2010) with such areas ideally placed where levels of biodiversity and the impact of threatening processes are high (Ricketts et al., 2005). However, PA placement is often constrained by additional anthropogenic motivations and may be biased towards low-value areas where the land cover clearing threat is relatively low (Joppa and Pfaff, 2010, 2011). More fundamentally, it can be difficult to evaluate the optimality of PA placement because we may value different facets of biodiversity in different ways and uncertainties in species distributions may not be accounted for. We investigated these issues for the bird order Galliformes in the Greater Himalaya, which has been identified as a priority region for conservation efforts with a disproportionately large number of threatened terrestrial species (Hoffmann et al., 2010) with current levels of deforestation likely to wipe out almost a quarter of endemic species, including 366 endemic vascular plant taxa and 35 endemic vertebrate taxa by 2100 in the Indian Himalaya alone (Pandit et al., 2006). We found there were some differences in the important conservation areas identified based on different ways of valuing biodiversity and that for three of the five different ways of prioritising biodiversity conservation that the configuration of PAs was significantly worse than the optimal solution.

Protection of at least 17% of terrestrial land is required by CBD targets (Secretariat of the Convention on Biological Diversity, 2010), but over 18% of the Greater Himalayan area is already protected, passing this target. However, we found evidence that the current PA network fails to adequately represent the distributions of Galliformes in the Greater Himalaya at the level of the CBD target for both un-weighted Zonation solutions and Zonation solutions weighted by IUCN Red List and endemism to the Himalayas. In contrast, we find that the current PA network reasonably represents the distributions of Galliformes weighted by phylogenetic diversity and relative range declines. The overlap between the top fractions of different facets of conservation value was small in terms of the overall proportion of the Greater Himalaya and only 23.5% of this combined value is already captured by the current Himalayan protected area network. This implies that different aspects of conservation value are represented somewhere within the current PA network, even if not necessarily in the same location and in sufficient amounts to achieve CBD targets.

We found species at the greatest risk of global extinction and those with the narrowest endemism to the Himalaya to be less well-covered by the current PA

network than the optimum solution which suggests re-designing the PA network would result in greater capture of the ranges of these species and so higher safeguarding of Himalayan Galliformes. This need not be based exclusively on the modelled data given here, but could incorporate locations identified as Important Bird Areas (IBAs; BirdLife International, 2008), which prioritises bird species with high global extinction risk and endemism among other criteria. Nevertheless, it is worrying that we are under-representing distributions for Galliform species threatened with global extinction in the current Himalayan PA network by 56% (average distribution of species covered across five different prioritisation methods – see Table 5.2).

Our study looked at representation and did not consider the efficacy of PAs in preventing direct and indirect extirpations. Effective enforcement is essential but Clark et al. (2013) showed that to date South East Asian PAs have not been successful at preventing habitat conversion. In addition, the results of Kaul and Ghose (2005) and Kaul et al. (2004) showed that the hunting of Galliformes is prevalent across the Himalayas. As such, more emphasis should also be placed upon management effectiveness and biodiversity outcomes as well as biodiversity representation. This is because, even if the location of the PA network did reasonably represent important Galliformes of conservation concern, it is unlikely that it offers much in the way of realised protection. Clark et al. (2013) suggested considering degazettement or realignment of PA boundaries in the most irreparably degraded areas and also advise that while difficult (Singh, 1985), it would be better to increase the quality and quantity of enforcement within PAs. We agree with the latter point but also caution that the re-alignment of PA boundaries should also take into account systematic conservation planning, so as to prevent any losses and potentially obtain gains of biodiversity representation. Similarly, greater enforcement would require further resources and it is established that PAs are underfunded (McCarthy et al., 2012). Therefore, the governments responsible for Himalayan PAs might need to allocate greater funding for PAs to be effective.

Our results come with some methodological caveats: the first caveat relates to our use of a species' environmental niche as a proxy for species occurrence. Thus, the realised niche of a species may depend on biogeographical, historical or biotic factors in addition to the abiotic factors used in our model, potentially increasing the unreliability of some of these models (Guisan and Thuiller, 2005; Rondinini et al., 2006). However, these areas should be more accurate than extent of occurrences, as used in previous analysis (e.g. Venter et al., 2014). We accounted for potential variations in model reliability by using a combination of distribution discounting and model weighting. Our results showed that accounting for variation in the niche modelling process did not change the

spatial outputs given by Zonation greatly. This is likely to be due to the use of the core-area algorithm rather than an artefact of the distribution discounting method itself. Nevertheless, by accounting for uncertainties in this way, we were able to ensure our results were as robust as possible.

The second caveat is related to the first, as by using niche modelling in conservation planning we run the risk of adding commission errors (false positives) to our analysis (Rondinini et al., 2006). If these commission errors are large, we might overestimate the true proportion of important conservation areas represented within the PA network (Rodrigues and Gaston, 2001). One way to account for this is to test our distribution models using field-based studies, though this may be impractical across such a large area. However, it is important to point out that by using the core-area algorithm in Zonation, we do retain areas of distributions with the highest probabilities of occurrence for each species (Moilanen et al., 2005; Moilanen, 2007), which reduces the likelihood of commission errors and that such errors may be small in comparison to other methods of representing species distributions such as extent of occurrence (Beresford et al., 2011; Rodrigues, 2011).

It is important to note, that while the current network does not adequately represent Galliform species of conservation concern at present, species distributions may move with climate change (Root et al., 2003), so the placement of PA network may be even more sub-optimal in the future (Hannah et al., 2007). This may have important ramifications, especially for alpine specialist species such as blood pheasant (*Ithaginis cruentus*), which may be trapped on effective 'sky islands' (Heald, 1967) making them particularly susceptible to climate change (Kupfer et al., 2005). Furthermore, we have grouped together different categories of PA, with different levels of protection afforded (Dudley, 2008) and that some of the PAs cross international boundaries with different managements regimes between countries. Therefore, in the future it may be necessary to investigate how climate changes may affect Galliform distributions, to assess the representation of Galliformes within different types of PA categories and for greater cross-border co-operation between Himalayan countries to occur if we are to ensure the continued survival of this bird Order.

Overall, our analysis provides new insights into the conservation of Galliformes within the Himalaya. We created new niche models for 23 out of 24 species of Galliformes, identified important areas for species of conservation concern using different types of conservation value and found that the current protected area network fails to capture these important localities for some types of value. We tested four different measures of species-specific conservation value relating to preserving biodiversity patterns and species

with the highest need of protection but other qualitatively different measures, such as preserving biodiversity processes and cost-effectiveness could be investigated further. In line with other studies, we suggest that conservation planners and legislators also need to devote more efforts to the problem of enforcement within Himalayan protected areas if we are to effectively prevent extirpations. Finally, although an important group from a conservation point of view, Galliformes represent a small fraction of the total biodiversity within the Himalayan region and it is important for future research to assess whether the results we observe are congruent with results for other taxa.

5.6 Appendix

5.6.1 Further information on the Maxent Species Distribution Modelling procedure

Dr Philip McGowan (Director of World Pheasant Association when work was conducted and co-author) and Dr Peter Garson (Chair of IUCN-SSC Galliformes Specialist Group) are Galliformes experts who were consulted for the selection of covariates for niche modelling.

Species	Model	Log likelihood	Parameters	Sample size	AIC	AIC_c	BIC
Blood pheasant	Maximal	-665.38	17	56	1364.75	1380.86	1399.18
Blood pheasant	Minimal*	-669.76	7	56	1353.52	1355.86	1367.70
Blyths tragopan	Maximal	-436.92	10	36	893.85	902.65	909.68
Blyths tragopan	Minimal*	-445.61	4	36	899.23	900.52	905.56
Buff-throated partridge	Maximal	-169.89	7	15	353.79	369.79	358.74
Buff-throated partridge	Minimal*	-173.53	4	15	355.06	359.06	357.89
Cheer pheasant	Maximal*	-5381.74	76	463	10915.49	10945.81	11229.96
Cheer pheasant	Minimal	-5518.48	28	463	11092.96	11096.70	11208.82
Chestnut-breasted partridge	Maximal	-321.85	10	27	663.70	677.45	676.66
Chestnut-breasted partridge	Minimal*	-323.17	4	27	654.34	656.16	659.52
Chukar	Maximal*	-1011.04	18	77	2058.07	2069.87	2100.26
Chukar	Minimal	-1033.30	6	77	2078.59	2079.79	2092.65
Common peafowl	Maximal	-815.03	16	65	1662.06	1673.39	1696.85
Common peafowl	Minimal*	-822.68	9	65	1663.36	1666.63	1682.93
Common quail	Maximal	-168.24	2	12	340.47	341.81	341.44

Table 5.A1: continued on next page

Species	Model	Log likelihood	Parameters	Sample size	AIC	AIC_c	BIC
Common quail	Minimal*	-168.24	2	12	340.47	341.81	341.44
Hill partridge	Maximal	-833.98	17	68	1701.95	1714.19	1739.69
Hill partridge	Minimal*	-849.52	5	68	1709.04	1710.01	1720.14
Himalayan monal	Maximal	-3825.92	79	303	7809.84	7866.53	8103.23
Himalayan monal	Minimal*	-3853.65	36	303	7779.31	7789.32	7913.00
Himalayan snowcock	Maximal	-763.98	20	62	1567.97	1588.45	1610.51
Himalayan snowcock	Minimal*	-780.05	6	62	1572.10	1573.63	1584.86
Kalij pheasant	Maximal	-2547.44	81	202	5256.88	5367.58	5524.85
Kalij pheasant	Minimal*	-2560.39	28	202	5176.79	5186.17	5269.42
Koklass pheasant	Maximal	-3476.97	72	275	7097.95	7149.99	7358.36
Koklass pheasant	Minimal*	-3533.47	29	275	7124.93	7132.04	7229.82
Red junglefowl	Maximal	-1435.46	41	116	2952.92	2999.46	3065.82
Red junglefowl	Minimal*	-1456.35	22	116	2956.70	2967.59	3017.28
Rufous-throated partridge	Maximal	-563.16	19	53	1164.32	1187.36	1201.76
Rufous-throated partridge	Minimal*	-571.86	14	53	1171.72	1182.77	1199.31
Satyr tragopan	Maximal*	-1542.65	23	132	3131.29	3141.51	3197.60

Table 5.A1: continued on next page

Species	Model	Log likelihood	Parameters	Sample size	AIC	AIC_c	BIC
Satyr tragopan	Minimal	-1559.15	13	132	3144.30	3147.38	3181.78
Sclaters monal	Maximal	-457.65	10	37	935.30	943.77	951.41
Sclaters monal	Minimal*	-458.88	6	37	929.77	932.57	939.43
Snow partridge	Maximal	-227.38	16	19	486.76	758.76	501.88
Snow partridge	Minimal*	-230.60	9	19	479.20	499.20	487.70
Tibetan eared pheasant	Maximal	-648.42	12	52	1320.83	1328.83	1344.25
Tibetan eared pheasant	Minimal*	-649.96	6	52	1311.93	1313.79	1323.63
Tibetan partridge	Maximal	-438.57	7	33	891.15	895.63	901.63
Tibetan partridge	Minimal*	-442.52	4	33	893.033	894.46	899.02
Tibetan snowcock	Maximal*	-1308.93	20	98	2657.86	2668.77	2709.56
Tibetan snowcock	Minimal*	-1329.70	8	98	2675.41	2677.02	2696.09
Temmincks tragopan	Maximal	-198.25	7	16	410.51	424.51	415.92
Temmincks tragopan	Minimal*	-198.59	5	16	407.18	413.18	411.04
Western tragopan	Maximal	-4072.90	91	350	8327.78	8392.68	8678.86
Western tragopan	Minimal*	-4168.45	22	350	8380.91	8384.00	8465.78

Table 5.A1: continued on next page

Species	Model	Log likelihood	Parameters	Sample size	AIC	AIC_c	BIC

Table 5.A1: The number of locality records used and selection of model (maximal or minimal) based on AIC_c. Asterisks (*) indicate the final model used. Key: AIC = Akaike Information Criterion, AIC_c = AIC corrected and BIC = Bayesian Information Criterion. AIC, AIC_c and BIC are all measures of the relative quality of each model. The Himalayan quail was included in Zonation as point locality data, so no Maxent modelling information is shown here.

	WWF ecoregion										
Species	**EHasm**	**EHb**	**EHsc**	**Hsb**	**Hsp**	**NEHsc**	**NTt**	**NWHasm**	**WHasm**	**WHb**	**WHsc**
Blood pheasant	10	9	27	4	0	3	0	0	0	1	0
Blyth's tragopan	4	17	4	0	0	2	1	0	0	0	0
Buff throated partridge	1	1	0	0	0	13	0	0	0	0	0
Cheer pheasant	1	0	2	2	122	0	0	33	36	146	113
Chestnut breasted partridge	3	10	8	0	1	1	0	0	0	0	0
Chukar	2	0	3	0	24	0	0	9	4	7	6
Common peafowl	0	0	0	10	18	0	0	0	0	5	1
Common quail	0	0	0	3	0	0	0	0	1	2	0
Hill partridge	3	24	22	1	11	1	0	1	0	4	1

Table 5.A2: continued on next page

	WWF ecoregion										
Species	**EHasm**	**EHb**	**EHsc**	**Hsb**	**Hsp**	**NEHsc**	**NTt**	**NWHasm**	**WHasm**	**WHb**	**WHsc**
Himalayan monal	19	16	24	3	45	5	0	67	17	58	44
Himalayan quail	0	0	0	0	4	0	0	0	0	0	0
Himalayan snowcock	1	0	2	0	8	0	0	14	7	10	11
Kalij pheasant	5	48	15	5	66	2	0	11	5	35	9
Koklass pheasant	2	0	27	0	71	3	0	54	7	68	29
Red junglefowl	1	32	1	9	44	1	1	6	2	8	10
Rufous throated partridge	0	33	8	1	11	0	0	0	0	0	0
Satyr tragopan	12	14	80	3	13	1	0	0	2	4	1
Sclater's monal	21	9	0	0	0	5	0	0	0	0	0

Table 5.A2: continued on next page

	WWF ecoregion										
Species	**EHasm**	**EHb**	**EHsc**	**Hsb**	**Hsp**	**NEHsc**	**NTt**	**NWHasm**	**WHasm**	**WHb**	**WHsc**
Snow partridge	4	0	2	0	0	0	0	5	5	3	0
Temminck's tragopan	2	7	2	0	0	3	0	0	0	0	0
Tibetan eared pheasant	0	0	0	0	0	8	0	0	0	0	0
Tibetan partridge	1	0	0	0	0	5	0	0	1	0	0
Tibetan snowcock	9	0	0	0	0	4	0	1	6	0	0
Western tragopan	0	0	0	0	21	0	0	99	9	100	110

Table 5.A2: The number of Galliformes locality records in each WWF ecoregion of our study site. Key: EHasm = Eastern Himalayan alpine shrub and meadows, EHb = Eastern Himalayan broadleaf forest, EHsc = Eastern Himalayan sub-alpine conifer forest, Hsb = Himalayan sub-tropical broadleaf forest, Hsp = Himalayan sub-tropical pine forest, NEHsc = North-Eastern Himalayan sub-alpine conifer forest, NTt = Northern Triangle temperate forest, NWHasm = North-western Himalayan alpine shrub and meadows, WHasm = Western Himalayan alpine shrub and meadows, WHb = Western Himalayan broadleaf forest and WHsc = Western Himalayan sub-alpine conifer forest.

Species	Site delimitation method	Feature function	Beta value	AUC	Standard deviation
Blood pheasant	SS	lq	0.5	0.862	0.058
Blyth's tragopan	ON	lq	0.5	0.815	0.105
Buff throated partridge	ER	lq	2	0.872	0.207
Cheer pheasant	ER	lqp	0.5	0.899	0.031
Chestnut breasted partridge	ON	lq	1	0.86	0.078
Chukar	ON	lq	0.5	0.855	0.035
Common peafowl	ON	lq	0.5	0.852	0.074
Common quail	ON	l	2	0.465	0.13
Hill partridge	SS	lq	0.5	0.845	0.075
Himalayan monal	SS	lqp	1	0.797	0.038
Himalayan snowcock	ON	lq	0.5	0.933	0.036
Kalij phesant	SS	lqp	1	0.799	0.036
Koklass pheasant	ON	lqp	1	0.921	0.02
Red junglefowl	SS	lqp	0.5	0.82	0.046
Rufous throated partridge	ER	lq	0.5	0.943	0.047
Satyr tragopan	ER	lq	0.5	0.879	0.053
Sclater's monal	ON	lq	0.5	0.806	0.073

Table 5.A3: continued on next page

Species	Site delimitation method	Feature function	Beta value	AUC	Standard deviation
Snow partridge	ER	lq	5	0.827	0.119
Temminck's tragopan	ON	lq	0.5	0.783	0.238
Tibetan eared pheasant	ON	lq	0.5	0.934	0.061
Tibetan partridge	ON	lq	1	0.803	0.12
Tibetan snowcock	ON	lq	0.5	0.798	0.115
Western tragopan	ON	lqp	1	0.95	0.009

Table 5.A3: Further details of Maxent models including Area Under Curve (AUC) and standard deviation outputs. Geographic extent delimitation method codes: SS = study site only, ON = occupied or neighbouring ecoregions and ER = Himalayan ecoregions only. Feature function (the shape of predictor response curves) codes: l = linear, q = quadratic and p = product. Beta = regularisation multiplier (how closely the output distribution is fitted to given presences). Note as Himalayan quail was included in Zonation using point locality, there is no Maxent model information to include here.

Species	Predictor
Blood pheasant	Oct NDVI, Dec NDVI, MAT, MAVT, MAP, MAVP, study site
Blyths tragopan	Jan NDVI, Feb NDVI, Mar NDVI, Oct NDVI, Dec NDVI, occupied neighbour
Buff-throated partridge	Jul NDVI, ecoregions
Cheer pheasant	Jan NDVI, Sep NDVI, MAVT, MAVP, elevation, slope, ecoregions
Chestnut-breasted partridge	Feb NDVI, May NDVI, Oct NDVI, Dec NDVI, MAT, occupied neighbour
Chukar	Apr NDVI, MAVT, MAVP, elevation, occupied neighbour
Common peafowl	Feb NDVI, March NDVI, Apr NDVI, May NDVI, Jul NDVI, MAVP, slope, occupied neighbour
Common quail	Jun NDVI, MAT, occupied neighbour
Hill partridge	Aug NDVI, Oct NDVI, Nov NDVI, Dec NDVI, MAT, MAVT, study site
Himalayan monal	Jan NDVI, Feb NDVI, Jun NDVI, Jul NDVI, Aug NDVI, MAT, MAVT, MAVP, elevation, slope, study site
Himalyan snowcock	Jun NDVI, MAVP, elevation, slope, occupied neighbour
Kalij pheasant	Mar NDVI, Apr NDVI, Jun NDVI, Sep NDVI, Nov NDVI, MAT, MAVT, MAP, MAVP, study site
Koklass pheasant	May NDVI, Jun NDVI, Jul NDVI, Aug NDVI, MAT, MAVT, MAVP, elevation, slope, occupied neighbour
Red junglefowl	Feb NDVI, Mar NDVI, May NDVI, Jul NDVI, Aug NDVI, Sep NDVI, Oct NDVI, Nov NDVI, MAT, MAP, MAVP, elev, study site
Rufous-throated partridge	Feb NDVI, Mar NDVI, May NDVI, Jul NDVI, Oct NDVI, Dec NDVI, MAT, MAP, ecoregions
Satyr tragopan	Apr NDVI, Jun NDVI, Oct NDVI, MAT, MAVT, elevation, ecoregions
Sclaters monal	Jul NDVI, Dec NDVI, MAT, MAVP, elevation, slope, occupied neighbour
Snow partridge	Feb NDVI, Mar NDVI, Apr NDVI, May NDVI, Dec NDVI, MAT, MAVP, elevation, ecoregions
Tibetan eared pheasant	Jul NDVI, MAT, MAVT, MAP, MAVP, slope, occupied neighbour
Tibetan partidge	Jul NDVI, MAT, MAP, occupied neighbour
Tibetan snowcock	Jul NDVI, Oct NDVI, MAT, MAVT, MAVP, occupied neighbour
Temmincks tragopan	Nov NDVI, Dec NDVI, MAT, MAVP, elevation, slope, occupied neighbour
Western tragopan	Jun NDVI, Jul NDVI, MAT, MAVT, MAP, slope, occupied neighbour

Table 5.A4: Further details of predictor variables used in final Maxent models. Key: MAT = Maximum Annual Temperature, MAVT = Maximum Annual Variability in Temperature, MAP = Maximum Annual Precipitation, MAVP = Maximum Annual Variability in Precipitation.

5.6.2 Maxent output maps

Figures 5.A1, 5.A2 and 5.A3 show the Maxent outputs used in the Zonation analysis.

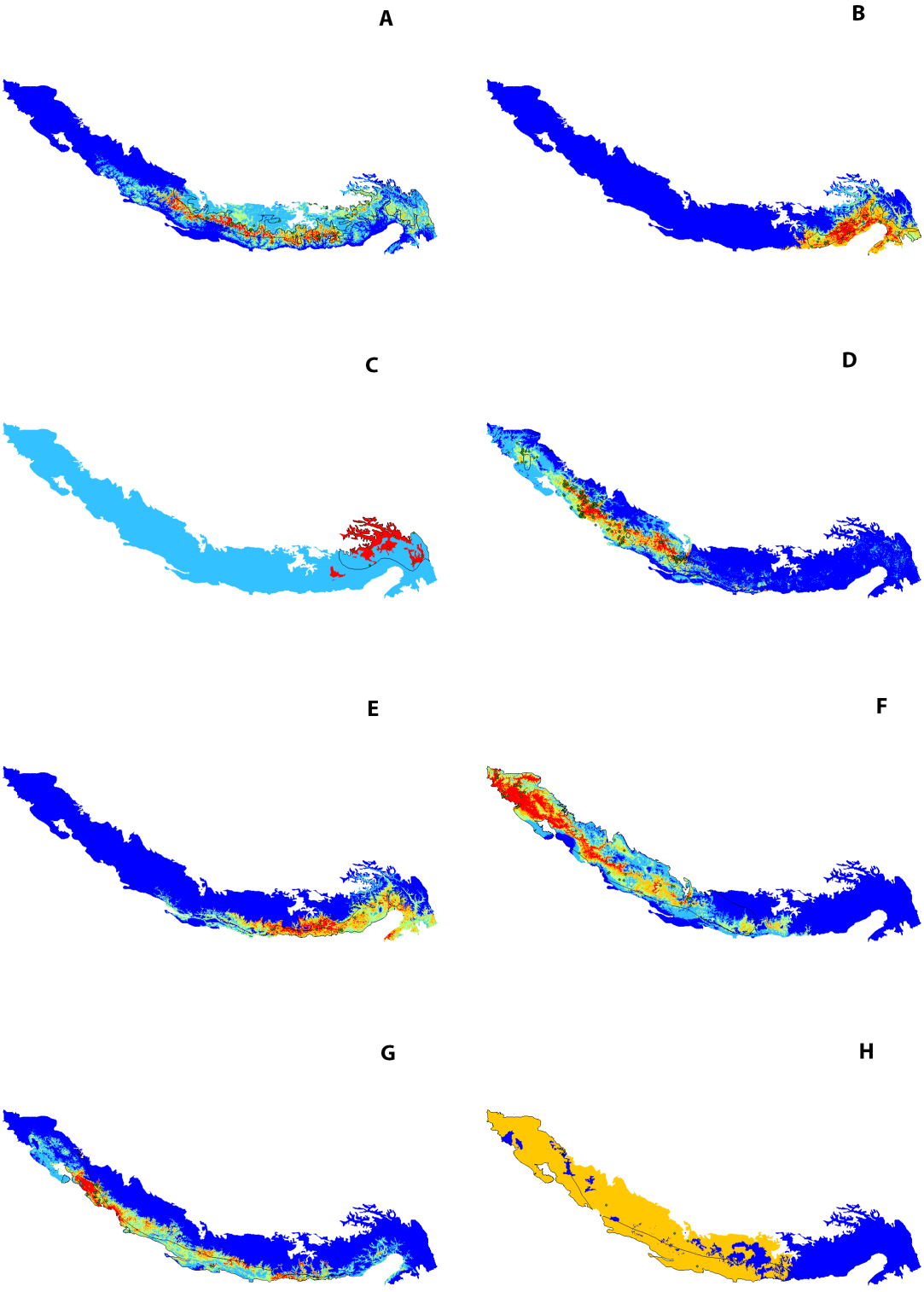


Figure 5.A1: Maxent output for 8/24 species part 1. Polygons = BirdLife shapefile, green dots = point localities and warmer colours = most suitable habitat and cool colours = least suitable habitat. Key: A = blood pheasant, B = blyth's tragopan, C = buff throated partridge, D = cheer pheasant, E = chestnut breasted partridge, F = chukar, G = common peafowl, H = common quail.

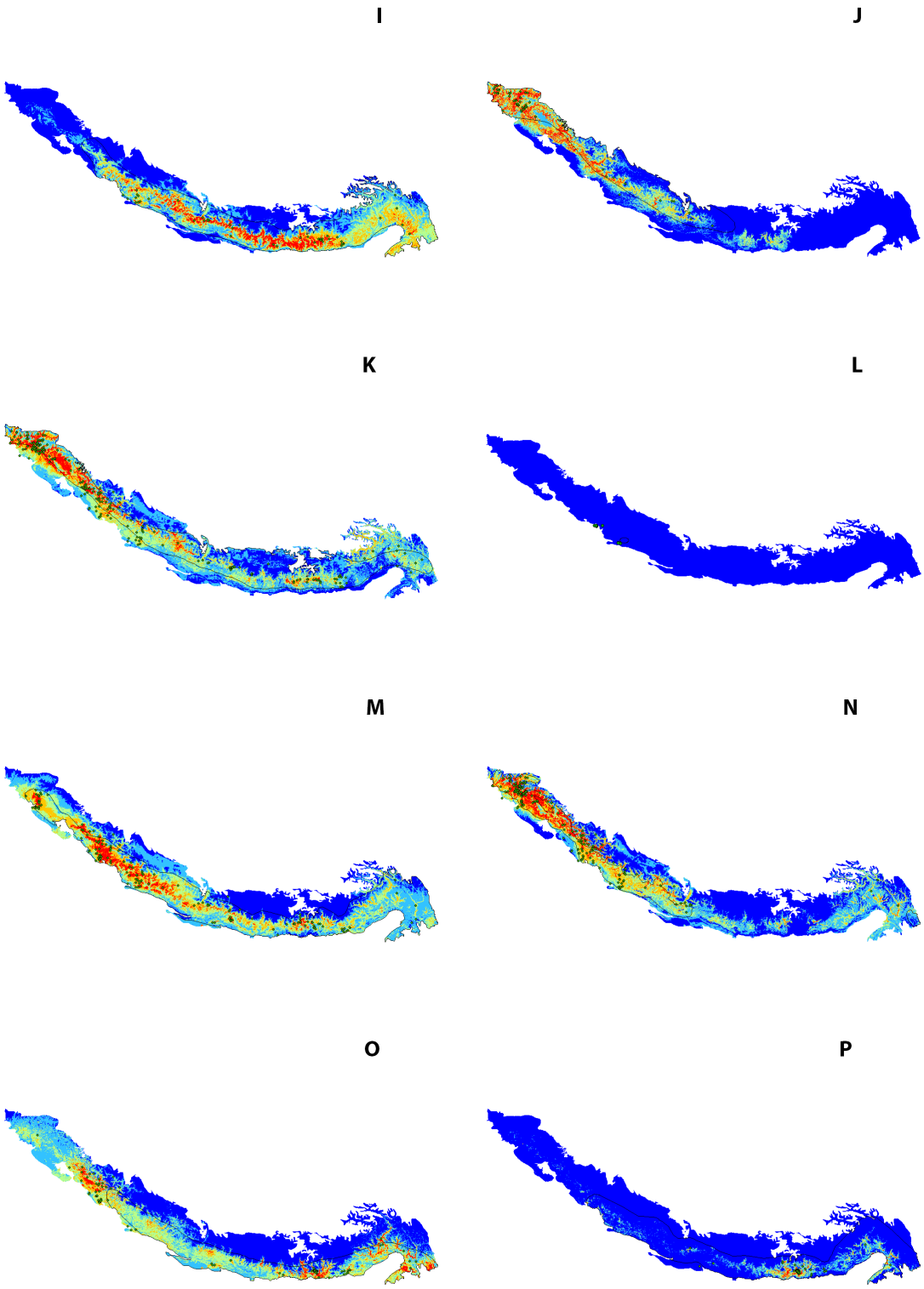


Figure 5.A2: Maxent output for 8/24 species part 2. Polygons = BirdLife shapefile, green dots = point localities and warmer colours = most suitable habitat and cool colours = least suitable habitat. Key: I = hill partridge, J = Himalayan snowcock, K = Himalayan monal, L = Himalayan quail, M = kalij pheasant, N = koklass pheasant, O = red junglefowl, P = rufous throated partridge. Note: Himalayan quail is shown as point data.

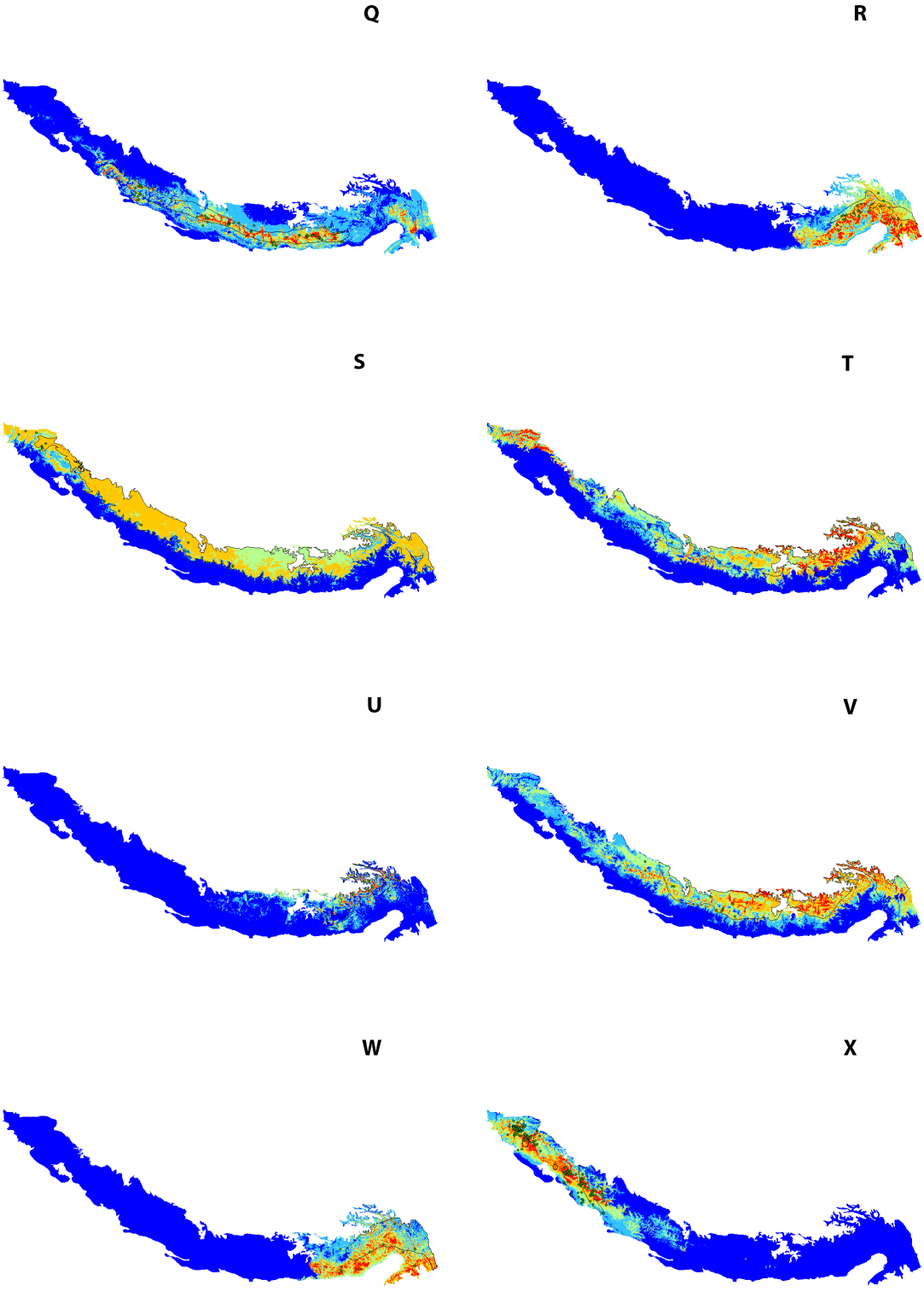


Figure 5.A3: Maxent output for 8/24 species part 3. Polygons = BirdLife shapefile, green dots = point localities and warmer colours = most suitable habitat and cool colours = least suitable habitat. Key: Q = satyr tragopan, R = sclater's monal, S = snow partridge, T = Tibetan partridge, U = Tibetan eared pheasant, V = Tibetan snowcock, W = temminck's tragopan, X = western tragopan.

5.6.3 Calculation of raw Zonation weights and Zonation output

The following section outlines details of species-specific conservation values. We described how scores were obtained and re-scaled.

- **Confidence in Maxent output**. Smooth response curves (Figure 5.A4) and maps that broadly agreed with expert opinion with AUCs greater than 0.5 (better than random) were scored between 1 (most confident) and 3 (least confident) to reflect our confidence in the Maxent niche modelling procedure. Examples of poor maps include the common quail (Figure 5.A1 panel H) and good maps the western tragopan (Figure 5.A3 panel X).
- **Red List**. Red List categories were taken from the IUCN Red List (IUCN, 2012) and converted to numerical scores (Least Concern = 1, Near Threatened = 2, Vulnerable = 3, Endangered = 4, Critically Endangered = 5). Thus, higher scores corresponded to a higher global extinction risk.
- **Phylogenetic distinctiveness (PD)**. Fair proportions scores were calculated by Will Stein using a new phylogeny. The method partitioned phylogenetic tree branches by the total number of species descending from them, regardless of nested tree structure, such that the contribution of a given ancestral branch to the PD score is 1/number of descendants of that branch. Analyses were conducted using all 24 species and a sensitivity analysis was undertaken by excluding chestnut breasted partridge (*Arborophilia mandelli*) and snow partridge (*Lerwa lerwa*), which lacked molecular data and were included in the phylogeny by genus-level constraints.
- **Relative range change**. Measures of range change were calculated after Telfer et al. (2002) – see Chapter 3 for further details. The standardised residuals represent range contractions and expansions relative to the trend of the group analysed and not absolute changes.
- **Endemism to the Himalaya**. Measures of endemism were calculated by working out the proportion of the Greater Himalaya that intersected the focal species' global range as measured by BirdLife range maps (BirdLife International and NatureServe, 2011). The endemism measure reflected the important of the Greater Himalaya as an areal proportion of each species' total global range. A continuous scale was used as definitions of endemism area inherently scale dependent (Laffan and Crisp, 2003) with 1 = 'endemic' and 0 = 'not endemic'.
- **Re-scaling the raw weight scores**. Scores were scaled to be congruent to the Red List categories (between 1-5) using the general formula (see equation 5.1) where mapping from (a,b) to (c,d). Note: the adjusted

weights for the relative range scores were reversed in polarity as species that have negative raw scores are higher priorities.

$$x' = \left(\frac{(x-a)(d-c)}{b-a} \right) + c \tag{5.1}$$

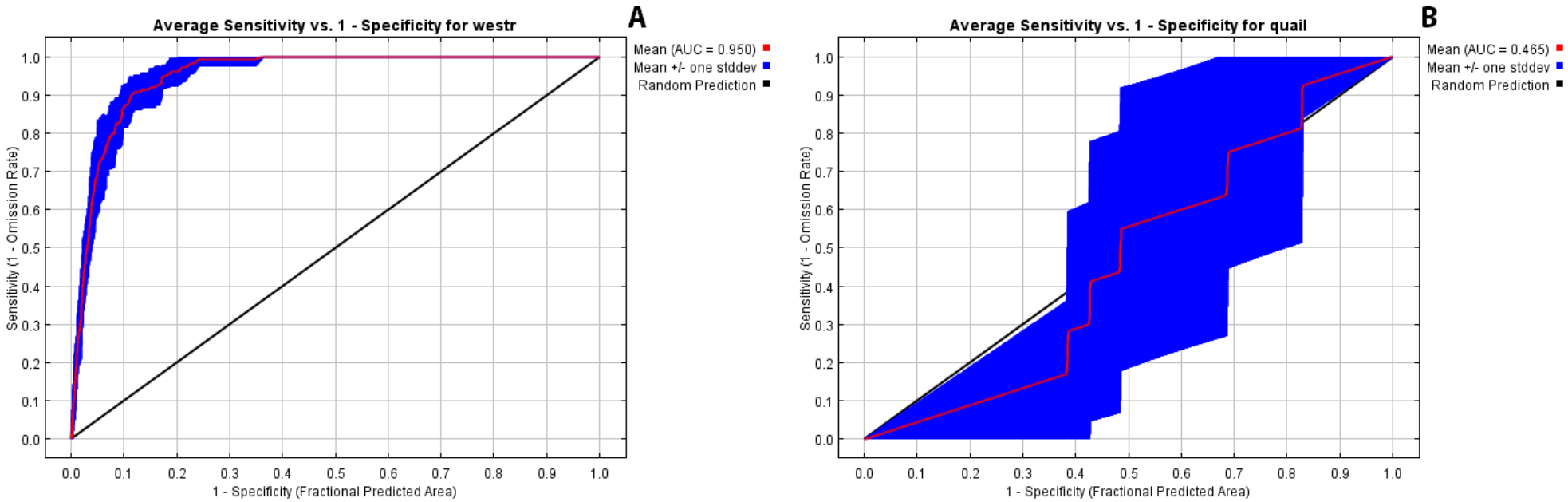


Figure 5.A4: Examples of 'good' (A) and 'bad' (B) Maxent receiver operator characteristics. A shows a smooth mean AUC with small standard deviations well above the random prediction line. B shows a jagged mean AUC with large standard deviations below the random prediction line.

Common name	Binomial	Confidence in Maxent output	Raw weights (adjusted weights)			
			Red List	Endemism	Relative Range Change	Phylogenetic distinctiveness
Blood pheasant	*Ithaginis cruentus*	2	1	0.22 (1.85)	-0.80 (3.17)	35.63 (5.00)
Blyth's tragopan	*Tragopan blythii*	2	3	0.23 (1.89)	1.20 (1.20)	10.43 (1.67)
Buff throated partridge	*Tetraophasis szechenyii*	3	1	0.21 (1.81)	N/A	14.15 (2.16)
Cheer pheasant	*Catreus wallechi*	1	3	0.74 (3.95)	0.67 (1.72)	11.74 (1.84)
Chestnut breasted partridge	*Arborophilia mandelli*	2	1	1.00 (5.00)	0.67 (1.72)	8.32 (1.39)
Chukar	*Alectoris chukar*	1	1	0.03 (1.08)	-0.59 (2.97)	8.18 (1.37)
Common peafowl	*Pavo cristatus*	1	1	0.12 (1.44)	0.82 (1.57)	15.07 (2.28)
Common quail	*Coturnix coturnix*	3	1	0.04 (1.12)	0.68 (1.71)	9.98 (1.61)
Hill partridge	*Arborophilia torqueola*	1	1	0.01 (1.00)	-0.09 (2.47)	8.11 (1.36)
Himalayan monal	*Lophophorus impejanus*	2	1	0.58 (3.30)	1.40 (1.00)	13.29 (2.04)
Himalayan quail	*Ophrysia superciliosa*	N/A	5	0.78 (4.11)	N/A	N/A
Himalayan snowcock	*Tetraogallus himalayensis*	1	1	1.00 (5.00)	0.34 (2.05)	10.28 (1.65)

Table 5.A5: continued on next page

Common name	Binomial	Confidence in Maxent output	Raw weights (adjusted weights)			
			Red List	Endemism	Relative Range Change	Phylogenetic distinctiveness
Kalij pheasant	*Lophura leucomenalos*	2	1	0.33 (2.29)	0.62 (1.77)	6.39 (1.13)
Koklass pheasant	*Pucrasia macrolopha*	1	1	0.10 (1.36)	1.19 (1.21)	24.83 (3.57)
Red junglefowl	*Gallus gallus*	2	1	0.05 (1.16)	0.71 (1.68)	9.58 (1.55)
Rufous throated partridge	*Arborophilia rufogularis*	1.5	1	0.26 (2.01)	-0.42 (2.80)	8.28 (1.38)
Satyr tragopan	*Tragopan satyra*	1	2	1.00 (5.00)	0.09 (2.29)	11.45 (1.80)
Sclater's monal	*Lophophorus sclateri*	2	3	0.55 (3.18)	-0.07 (2.45)	13.29 (2.04)
Snow partridge	*Lerwa lerwa*	2	1	0.4 (2.58)	-0.21 (2.59)	23.39 (3.38)
Temminck's tragopan	*Tragopan temminckii*	3	1	0.06 (1.20)	0.01 (2.37)	10.97 (1.74)
Tibetan eared pheasant	*Crossoptilon harmani*	1	2	0.62 (3.46)	-1.47 (3.83)	5.39 (1.00)
Tibetan partridge	*Perdix hodgsoniae*	2	1	0.17 (1.65)	-2.65 (5.00)	13.37 (2.06)
Tibetan snowcock	*Tetraogallus tibetanus*	2	1	0.17 (1.65)	0.05 (2.33)	12.07 (1.88)

Table 5.A5: continued on next page

Common name	Binomial	Confidence in Maxent output	Raw weights (adjusted weights)			
			Red List	Endemism	Relative Range Change	Phylogenetic distinctiveness
Western tragopan	*Tragopan melanocephalus*	1	3	0.78 (4.11)	1.20 (4.11)	10.481.67)

Table 5.A5: Summary of the species used in the analysis with details of species-specific weightings. Note: the adjusted weights for the relative range change scores have been reversed in polarity as species that have negative raw scores are higher priorities. Confidence in Maxent outputs were assessed by G. Buchanan and J. Dunn.

5.6.4 *Further information on Zonation output overlap*

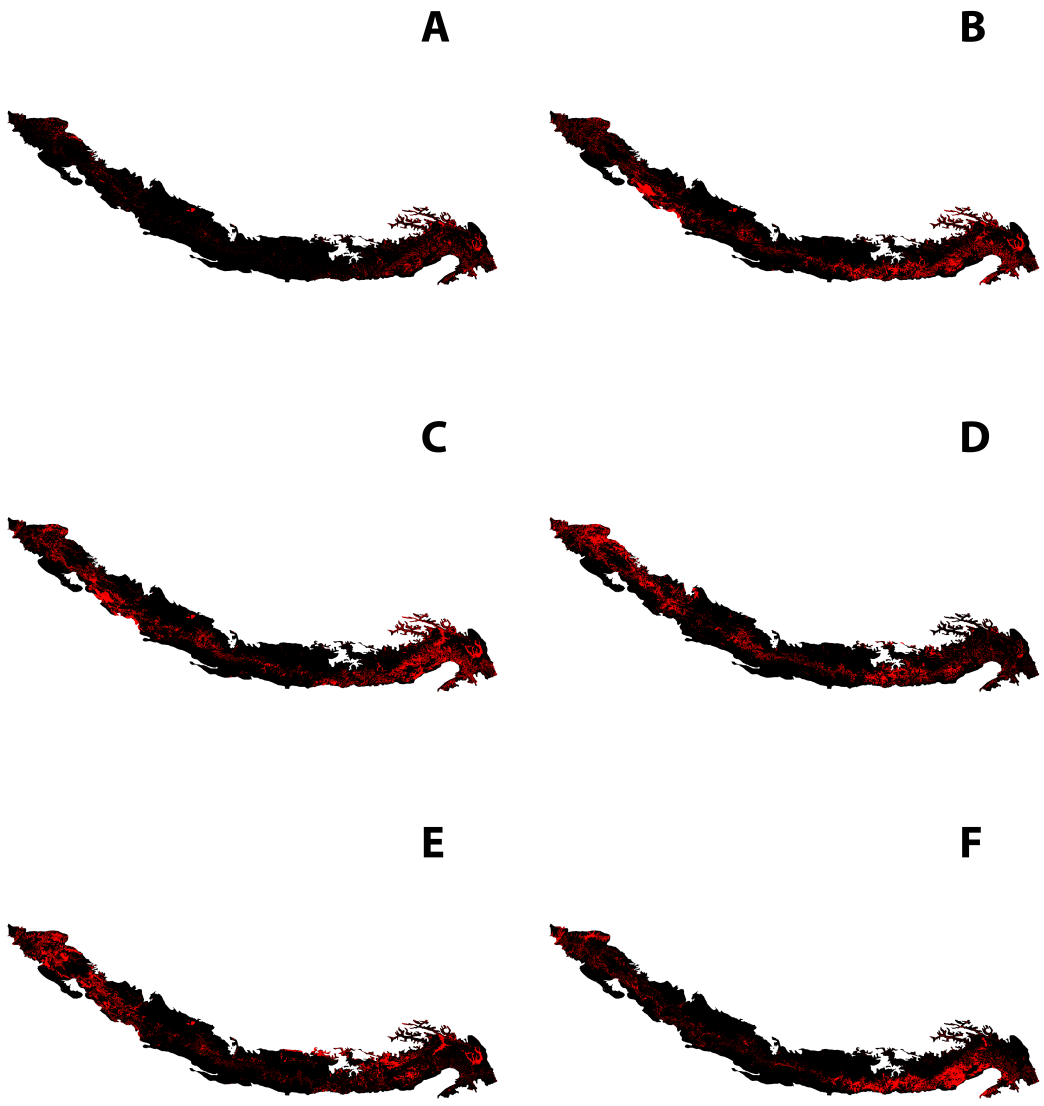


Figure 5.A5: Comparison maps for Kappa statistics part 1. Black indicates areas with equal value rasters (0+ 0 or 1+1) whereas red indicates areas with unequal value rasters (0+1). Total no. of cells in area = 687,532. Overlaps are shown for: A = basic x distribution discounting, B = distribution discounting x endemism, C = distribution discounting x Red List, D = distribution discounting x phylogenetic diversity, E = distribution discounting x relative range declines, F = endemism x relative range declines.

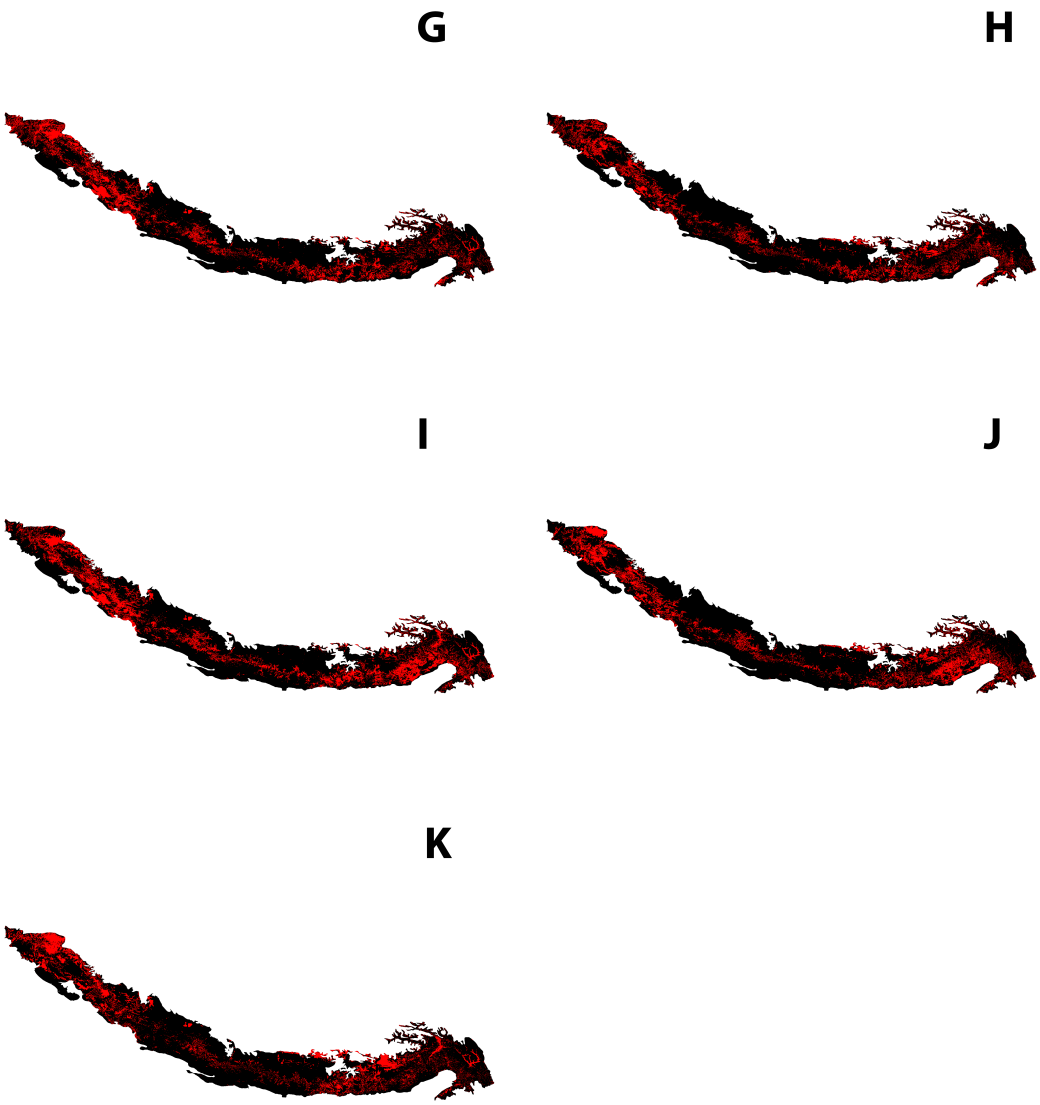


Figure 5.A6: Comparison maps for Kappa statistics part 2. Black indicates areas with equal value rasters (0+0 or 1+1) whereas red indicates areas with unequal value rasters (0 and 1). Total no. of cells in area = 687,532. Overlaps are shown for: G = endemism x phylogenetic diversity, H = endemism x relative range declines, I = Red List x phylogenetic diversity, J = Red List x relative range declines, K = phylogenetic diversity x relative range declines.

Chapter 6. Discussion

Global biodiversity is currently undergoing large losses (Butchart et al., 2010; Pimm et al., 2014). The impacts arising from this loss are likely to be large (Millennium Ecosystem Assessment, 2005), both for biodiversity itself and humanity, with major disruptions to ecosystem services and losses in natural capital (TEEB, 2010). As a result, there are political commitments to reduce biodiversity loss by the year 2020 (Secretariat of the Convention on Biological Diversity, 2010). Conservation will play an important role in achieving these commitments at least in part, by directing limited resources to priority species and sites. Deciding which species and which sites should be priorities can be challenging and is complicated by data shortfalls (e.g. Kozlowski, 2008). In this thesis I have focused on one region and one particular taxonomic group: the avian order Galliformes that live within the Greater Himalaya. The same conservation questions apply: which species and which sites should we prioritise conservation resources towards? To help answer these conservation questions I have examined both aspects of species declines and the distributions of Himalayan Galliformes with the aims to: (1) map current distributions of Himalayan Galliformes (Chapters 2 and 5); (2) map past geographic ranges and compare them with more contemporary distributions to explore patterns of long-term range changes (Chapter 3); (3) compare global range changes of narrowly distributed and widespread species (Chapter 4); (4) identify whether current spatial conservation measures are optimal for Himalayan Galliformes (Chapter 5).

Overall, my thesis provides new evidence and insights for the conservation of Himalayan Galliformes but also acts as a microcosm of global conservation issues. In this discussion I will first re-iterate the main findings from my data Chapters in relation to my four original aims; then I will discuss how my findings can help direct conservation resources to priority species and sites within the Himalaya; next, I will recommend future research directions for the conservation of Himalayan Galliformes; and finally, I will discuss how my research can help in a broader sense to achieve global biodiversity targets.

6.1 Main findings of thesis

6.1.1 Aim 1: to map current distributions of Himalayan Galliformes

In Chapter 2, I examined the conservation status and distribution of one Galliform: the Himalayan quail (*Ophrysia superciliosa*). The Himalayan quail is listed by the IUCN as Critically Endangered and has not been reliably recorded since 1876. Recent searches have failed to detect the species and there are concerns that it may be extinct. I found that there is still considerable uncertainty regarding the Himalayan quail's extinction status but my calculations suggested that it may still be extant. It is therefore possible that searches have been conducted in the wrong places. Species distribution models (SDMs) can act as a guide for survey efforts, but the relationship between current land cover and historical specimen records is unlikely to reflect Himalayan quail habitat preferences due to extensive modifications to land use. I created a novel method to obtain greater predictive insight into the geographic distribution of the Himalayan quail by using the habitat preferences of two proxy species and measures of search effort based on presence-only records from other Galliform species. This allowed identifications of new potentially suitable areas to target future searches for the quail. My results will help to gain further insight into both the extinction status and distribution of this endangered bird species.

In Chapter 5, I mapped the current distributions of all other Himalayan Galliformes in addition to the Himalayan quail as part of my efforts to answer Aim 4 (whether current spatial conservation measures are optimal for Himalayan Galliformes).

6.1.2 Aim 2: to map past geographic ranges and compare them with more contemporary distributions to explore patterns of long-term range changes

Identifying and measuring species declines is a key component of conservation biology and the IUCN Red List method of categorising species by extinction risk is a widely accepted approach to assessing such changes. However, the Red List is typically concerned with short time frames and global spatial scales, which means that long-term and sub-global declines may go undetected and, therefore, not accord with Red List categories. Given that these categories are often used for conservation baselines and priority-setting, it is helpful to examine species declines across different spatio-temporal scales so as to make both informed and practical conservation decisions for Galliformes within the Himalaya. For example, examining which species have undergone long-term but pervasive range declines is an example of 'preventive conservation' (see

Chapter 1 section 1.3), because these species, while not listed as threatened by the Red List now, could potentially respond better to conservation efforts in comparison to threatened species.

In Chapter 3 I used long-term measures of geographic range change to assess species declines or declines in conservation status both inside and outside the Himalaya. Within the Himalaya, nine Galliform species were found to have declined in range size in relative terms, seven had expanded and six had remained relatively stable. I found that long-term range changes for Galliformes within the Himalaya were similar to global patterns. The proportion of a species' global range that intersected the Himalaya was also important to consider, given that for some globally widespread species the Himalaya was a very small part of their global range. However, when endemism to the Himalaya was accounted for, I still found broadly similar trends between the resulting conservation priorities. For my suite of species, the conservation priorities given by contemporary measures of extinction risk (i.e. Red List category) did not accord with those given by long-term range changes. It is not clear what has driven this discrepancy, but an important consequence is that if Red List categories are used to inform/set regional conservation priorities in the Himalaya, species that are categorised as non-threatened ('Least Concern' or 'Near Threatened') but have undergone large historical declines in conservation status are likely to be under-represented. I identified Tibetan eared-pheasant (*Crossoptilon harmani*) and Tibetan partridge (*Perdix hodgesoniae*) as species that have undergone large declines in conservation status within the Himalaya, which makes up a large part of their global geographic range but are currently listed as non-threatened by the IUCN Red List. These are species that I feel require a review of their current conservation status within the Himalaya and may require further consideration in regional conservation plans. Thus, in this Chapter I have gained further insight into the conservation status of Galliformes in the Himalaya by examining long-term changes in geographic range. Even though examining relative range changes and declines in conservation status over sub-global scales and long time periods can be challenging due to issues of data availability, I have shown that it is important to investigate new methods of doing so.

6.1.3 Aim 3: to compare global range changes of narrowly distributed and widespread species

Geographically widespread species play an important role on our planet, driving both biodiversity patterns and ecosystem functioning, but concern exists that we are overlooking long-term, marked declines in such species.

Conservation biology is generally concerned with narrowly distributed species, as a small range size has been shown to be the single biggest predictor of extinction and is easier to monitor and track the status of such species over small time periods. Thus, adjustments to the ways we assess species declines in widespread species could be needed in order to provide a more informed and 'preventive' approach to their conservation. This applies both to Galliformes in the Himalaya but also to biodiversity more generally.

In Chapter 4, I examined long-term range changes across all Galliformes in the world to put this issue in context. The results showed that there were no differences in the rates of long-term range declines between historically restricted-range species and widespread species. There was also no consistent pattern between these historical declines, Red List category and historical range size. This suggests implicit support for the Red List framework with no special adjustments to Red List procedures needing to be made when assessing range declines in widespread species. Although these results apply for all Galliformes across the globe, it also seems to suggest that geographically widespread Himalayan Galliformes in general do not need to be given a greater conservation focus than they presently receive.

6.1.4 Aim 4: to identify whether current spatial conservation measures are optimal for Himalayan Galliformes

Protected areas (PAs) are a key tool in protecting biodiversity but optimising their placement in the landscape is challenging due to a poor knowledge of biodiversity coverage and differences in how different aspects of biodiversity are valued. I assessed these two issues for Himalayan Galliformes in Chapter 5. To do this I compared the current location of the Himalayan PA network to important areas for Galliformes as identified from Zonation prioritisation software and modelled species distributions. Specifically, I identified areas of high species richness and then weighted maps by different species-specific conservation values including Red List score, a measure of regional relative declines, endemism to the Himalaya and phylogenetic distinctiveness. I found that regardless of the type of conservation value examined, the North West of India, Central Bhutan and the North East/South China border were consistently important for Himalayan Galliformes. I found that statistically significantly fewer important areas for Galliformes were represented in the actual Himalayan PA network in comparison to the optimal network (based on three different weighting schemes: Red List category, endemism to the Himalayan region and an un-weighted analysis that treated each species with equal importance). As a consequence, I suggested that some refinements to the current PA network are needed in order to maximise its effectiveness for

Galliformes, which might be based on BirdLife Important Bird Areas or Endemic Bird Areas. These results help contribute towards area-based efforts to conserve Himalayan Galliformes.

6.2 Synthesis: how my thesis helps to conserve Galliformes in the Himalaya

In order to ensure my thesis has a practical legacy for stakeholders and policy-makers in the Himalaya, in this section I discuss how my findings contribute towards the conservation of Himalayan Galliformes. This will be achieved by discussing the links between my data Chapters to achieve a synthesis of my results. I structure this discussion around two broad conservation questions: which species and which sites should we direct conservation resources towards? My thesis can help provide some of the information required for a regional strategic plan or Species Conservation Strategy (SCS; IUCN/SSC, 2008) for the conservation of Galliformes in the Himalaya and I outline where this can be achieved. Finally, I identify and discuss future research directions for the conservation of Himalayan Galliformes, which I then summarise as a list of recommended next actions.

6.2.1 Which species to direct conservation resources towards?

Obtaining a definitive answer of which species require the most conservation attention for the purposes of monitoring programmes, policy and direct action is difficult. The answer depends in part on the desired future state for the suite of species in question, or what is sometimes called the 'vision' in strategic planning terminology (IUCN/SSC, 2008). In my thesis, I have mainly examined aspects of species-level priority-setting that have focused on vulnerability to the risk of global extinction within the 'irreplaceability/vulnerability' framework (Brooks et al., 2006) that I introduced in Chapter 1. Conservation approaches that center around vulnerability may also be further sub-divided into those that are 'preventive' or 'proactive' and those that are 'fire-fighting' or 'reactive' (Brooks et al., 2006; Wilson et al., 2011).

If the conservation vision of policy-makers and planners is to avoid imminent extinctions in a reactive approach, the top conservation priority should be the species listed as threatened by the IUCN Red List, such as the Himalayan quail, but prioritizing attention towards these species may paradoxically not minimise extinctions. In the case of the Himalayan quail, if continued searches fail to detect the species, then further conservation investment may not be the best use of limited conservation resources (e.g. Bottrill et al., 2008). Even if it is

found that the quail is still extant, its long-term survival may be contingent on continued funding, which could arguably be used to minimise species extinctions elsewhere in the Himalaya. Cost-effectiveness is not an issue that I have examined in this thesis, but may have bearings on how both Himalayan Galliformes and biodiversity in general is conserved. I discuss this further in section 3 of this Chapter. Nevertheless, given its poor conservation status I feel it should be a top conservation priority for the foreseeable future.

If a more preventive view of prioritizing conservation action between species is the vision, then long-term range changes including my results from Chapters 3 and 4 can be used to give a more complete picture of contemporary conservation status and could help to minimise species extinctions for Himalayan Galliformes overall. In Chapter 3 I found that conservation priorities given by current Red List categories did not accord with those given by long-term range changes. As more narrowly distributed species tend to be listed as threatened on the Red List, this result could have been indicative of a broader trend of widespread species undergoing more persistent and pervasive relative range declines but at rates that have not triggered a threat categorization on the IUCN Red List. Chapter 4 showed that this is unlikely to be true. If these results are taken together, it is unlikely that all globally widespread species that are found in the Himalaya are in need of extra conservation attention purely as a function of them being widespread per se. However, some species for which the Himalaya is a large part of their global range, have undergone large declines in conservation status and these are the species that I feel Himalayan monitoring programmes and conservation policy may need to better consider in addition to the Himalayan quail. Nevertheless, it is interesting that for both Chapters 3 and 4, the species that had undergone the greatest long-term range declines (relative or absolute) were not always those listed as most threatened with extinction by the IUCN Red List, further supporting the idea that the Red List should not be used to set species-level priorities on its own (Mace and Lande, 1991; Lamoreux et al., 2003; Rodrigues et al., 2006).

While I have mainly focused on species-level prioritisation approaches that pertain to vulnerability (as these relate to Aichi Target 12; see Chapter 1) there are also conservation approaches that focus on irreplaceability as measured through endemism and phylogenetic distinctiveness. Phylogenetic distinctiveness (PD) scores were calculated as part of Chapter 5, but were used in a site-based priority-setting analysis, rather than a species-level priority-setting analysis. These PD scores could be used and combined with vulnerability scores (i.e. Red List categories) as have been done by the ZSL's 'Edge of Existence' project (Isaac et al., 2007). The Himalayan quail is the most

phylogenetically distinctive of the Himalayan Galliformes and is the only member of its genus. Thus, it could be argued that if I had focused on irreplaceable as well as vulnerable species, the Himalayan quail would still be a top conservation priority. Another way that I have indirectly examined species irreplaceability in priority-setting was in Chapter 3, where I combined measures of endemism to the Himalaya with measures of species range change, although this was more to do with the proportion of species' global range contained within the Himalaya, rather than absolute measures of range size. While this is not something that I have sought to examine, there are ways of combining multiple measures of vulnerability and irreplaceability, such as the example provided by Keith et al. (2007), which could be used to assess plurality in conservation priorities for Himalayan Galliformes.

In terms of usefully contributing towards a SCS for Himalayan Galliformes, my results could form part of a regional status review. Specifically, Chapter 2 helps to provide information on the current distribution and status of the Himalayan quail and Chapter 3 helps provide information on the historical distribution of Himalayan Galliforms and how their ranges have changed.

6.2.2 Which sites to direct conservation resources towards?

Obtaining a definitive answer to this question again relates to the conservation vision of Himalayan policy-makers and planners. Aichi Target 11 (see Chapter 1) states that conservationists should protect areas that are '…of particular importance for biodiversity' (CBD, 2010). Defining 'importance' is difficult, but implicit in Target 11 is the idea that directing resources to priority sites depends in part on which species are identified as conservation priorities. I illustrate this point in Chapter 5, using my range change results from Chapter 3 with other species-specific attributes in a spatial priority-setting context. As those results have showed that the placement of the current PA network did not represent Galliformes distributions well, solid recommendations need to be made regarding where the network should be expanded or replaced in the case of under-performing PAs (Fuller et al., 2010). While I have recommended that expanding the PA network based on Important Bird Areas and Endemic Bird Areas could be undertaken, this would require the spatial congruence between Galliform distributions and those for other taxonomic groups to be considered. If there were spatial congruence, Galliformes could also potentially be used as umbrella species or flagships for the conservation of biodiversity more generally in the Himalaya.

My thesis also indirectly helps to prioritise spatial conservation efforts by providing updated information on the current distributions of Himalayan

Galliformes in Chapters 2 and 5, which are usually needed before conservation resources can be directed towards certain sites. In this sense my results provide much utility for a regional SCS. My distribution maps take the form of species distribution models that show a continuous probability of habitat suitability. This is in contrast to the maps currently available (i.e. BirdLife range maps; BirdLife International and NatureServe, 2011), which take the form of a binary 'suitable' vs. 'unsuitable' output. The type of conservation resources or effort that I have discussed in relation to these distributions differs between the Chapters: in Chapter 2, I have provided a list of sites that should be surveyed for the Himalayan quail but in Chapter 5, I have provided maps to help re-design and optimise the placement of the current PA network for Himalayan Galliformes.

Nevertheless, the maps I have created in Chapter 5 could have multiple potential uses that relate to prioritising conservation resources between sites. For example, they could act as guides for surveys; to verify local extinctions/colonisations in conjunction with point locality data using a similar method to that used in Chapter 2; to test range limits; to identify key sites for long-term monitoring; and to identify habitat/resource use and thus, opportunities for habitat restoration. Many of these uses would be helpful in developing an SCS. In addition to surveying for Himalayan quail, I propose that Tibetan eared pheasant and Tibetan partridge are targeted for surveys first, as these species have undergone large, continuing declines in conservation status but are not listed as threatened by the Red List. The utility of these maps has already been proven as I have helped the Galliformes Specialist Group verify whether trip reports of cheer pheasant (*Catreus wallichii*) in Shey-Phoksundo National Park, Nepal were likely to be reliable based on my results.

6.2.3 Future research directions and recommendations

My results have provided information on which species and which sites should be considered current conservation priorities for Himalayan Galliformes and thus, many aspects that could be used for a regional level status review of this group within an SCS framework. For example, my thesis has provided information on historical and current distributions but also species declines (in relative or absolute terms). However, there are still a number of areas that require further research, both in terms of an SCS and also for the conservation of Galliformes in the Himalaya more generally. More generally, there is a need to better move beyond defining problems to identifying the nature of solutions, which will need to account for human population growth, livelihoods equity, ecosystem services, climate change and emergent threats. I outline the issues

that I feel are the most important below before summarizing them in a list of recommendations at the end of this subsection.

Drivers of species declines

First and foremost, it would be helpful to conduct a more explicit threat analysis (e.g. using methods outlined by Caughley and Gunn, 1996) to identify the drivers of species decline (both relative and absolute) in Chapters 3 and 4, because if the drivers of declines in conservation status were known it could help identify how best to ameliorate those drivers. Identifying which drivers have caused long-term declines in conservation status is difficult because linking changes in species populations or range sizes to changes in drivers requires information on both. This is problematic as data availability is often uneven or lacking. For example, in terms of drivers hunting is thought to be an important cause of species declines in Himalayan Galliformes (e.g. Kaul et al., 2004) but obtaining data on hunting is extremely difficult (Gavin et al., 2010). Habitat loss is another important driver of decline, but is easier to quantify, with the HYDE dataset providing one possible way of examining historical habitat changes (e.g. Clark et al., 2013). Another threat that may be important is forest fire (Kimothi and Jadhav, 1998), which could be usefully modelled in the future to predict which areas are most at risk (e.g. Gonzalez et al., 2006). Hydroelectric power also been identified as an important but understudied threat to Himalayan biodiversity (Grumbine and Pandit, 2013) and more field-based case studies in Uttar Pradesh, India could help investigate its effects on Galliformes further.

Patterns of species decline

Second, it may be helpful to examine patterns of species declines such as extinction dynamics or range fragmentation in more detail. The way ranges collapse could help inform our conservation responses as to which species and which sites require action (Channell and Lomolino, 2000). Thus, we may be able to target places where threatening processes are high (Channell and Lomolino, 2001) or to emphasise the importance of areas of species' range where persistence is high in systematic conservation planning. Similarly, where possible, it would be useful to examine range changes over a finer temporal scale, which would help better to link declines to threats.

Units of biodiversity

Next, it could also be useful to look at other aspects of Galliformes biodiversity, such as genetics and populations, which could be examined in relation to range

changes. Direct population abundance measures could also be used rather than proxies such as geographic range, which I used in Chapters 3 and 4 because there was a limited amount of spatial and population data available. It could be informative to calculate measures of occupancy for each locality across the Himalaya (e.g. Karanth et al., 2010) or to use direct measures of abundance that have been corrected for search effort (e.g. the FRESCALO technique; Hill, 2012; Isaac et al., 2014), which could help provide further information on patterns of species declines in conservation status. One innovative approach that could be used is to examine birder's lists to estimate population trends (Szabo et al., 2010).

Future-proofing conservation responses

While I have examined which species and sites are current conservation priorities, it is also important to identify which Galliform species and which Himalayan sites should be our future conservation priorities. For the former, this can be done by predicting which threats are likely to be important in the future and identifying species that are susceptible to such threats or those that have a high latent risk of extinction (e.g. Cardillo et al., 2006). It could also be useful to look at the functional responses of different species in relation to different threats. For example, the degree of habitat specialisation and the size of a species' elevational range has been shown to increase the extinction risk for tropical montane bird species in the face of global climate change (Sekercioglu et al., 2012). Similarly, given that many Himalayan habitats are likely to be converted for agriculture in the future (Pandit et al., 2006), widespread and generalist granivorous Galliform species may do better under such conditions (Maas et al., 2009).

In this sense, it might be interesting to also examine what the characteristics of the 'winners' are in Chapter 3, as the functional diversity of different Galliform species could help us to target conservation responses. Similarly, it may be necessary to identify future priority sites as climate change may cause range shifts in many species (e.g. Chen et al., 2011) and especially high altitude alpine obligates (La Sorte and Jetz, 2010). This means that the expansion of the PA network that I have advocated in Chapter 5 may need to also account for future species distributions under different climate change scenarios. Incorporating home range size estimates into our systematic planning framework would also help with this by accounting for connectivity better.

Political units

Recognising that countries are the spheres in which conservation action will take place is extremely important. Within my study region there exists a large disparity in both the absolute size of various countries and their economies, the responsibility they bear towards Himalayan biodiversity and attitudes towards conservation/wildlife (see Table A.1 in the thesis Appendix for further details). These issues could act as barriers to diverting priority-funds between different countries and will require further consideration although I have not examined them in detail within my thesis.

First, some Himalayan countries have been shown to underinvest in conservation resources relative to the global country average. For some such as Bhutan (23rd most underfunded country out of 198 assessed), this is a function of their small size, but there are also others such as China (39th most underfunded country out of 198 assessed), where the cause is less clear (Waldron et al., 2013). This has manifested itself in a disparity in both capacity to undertake conservation in Himalayan countries, but also in governance, which has been shown to be important in implementing conservation (Smith et al. 2003). Second, different countries bear differing levels of responsibility towards the conservation of Himalayan Galliformes: arguably the countries with most responsibility should be those that take up the largest area within the Himalayan region. This could have practical implications on the efficacy of the PA network among other things. For example, 98% of PAs within the Himalayas cross multiple borders (see Chapter 5) and different types of PA afford different levels of protection. All of these things could affect enforcement, which is crucial to conservation success. Finally, different countries have and have had different attitudes towards biodiversity conservation and science. This has manifested itself in, for example, patterns of record collection and as such, it might be worthwhile to re-examine some of the patterns that I have found using country as a random effect. In a practical sense, there are still differences in how each Himalayan country undertakes field surveys. Differing attitudes to conservation between countries also means that other methods of affording spatial conservation protection may be needed. For example, research in Nepal has identified that more community participation should be a part of conservation activities and has highlighted that community forests may have high conservation value even if they are without formal protection, generating livelihood opportunities that decrease pressure put on PAs (Dahal et al., 2014).

Recommendations

Based on my findings and on the arguments outlined above, I make the following recommendations:

1. Conduct searches and surveys for the Himalayan quail, Tibetan partridge and Tibetan eared pheasant.
2. Undertake a region-wide threat analysis including more case studies at local level that incorporate social science approaches (e.g. hunting) and identify where there may be mismatches in the types of conservation responses, if any, for each threat.
3. Investigate patterns of range change and proxies of abundance over different temporal scales and in relation to other facets of Galliformes biodiversity, such as genes and populations.
4. Investigate which species and which areas are likely to be future conservation priorities within the Himalaya based on emergent threats, latent extinction risk and functional responses to threats.
5. Review ecological census techniques used in each Himalayan country/ Galliform species to help standardise methodologies, search effort and help capacity building in countries that most need it. Ensure there are consistent between-country monitoring programmes and protocols for effective collaboration, especially between shared PAs.
6. Conduct field surveys using SDMs to test for range limits, local extinction/colonisation, habitat associations and opportunities for habitat restoration.
7. Identify if other taxonomic groups are spatially congruent with Galliformes, as they could act as umbrella species and flagships for conservation.

6.3 How my thesis can help achieve global biodiversity targets

As I outlined in Chapter 1, the earth is currently in the middle of a biodiversity crisis (Millennium Ecosystem Assessment, 2005). The implications of this crisis have been recognised with 168 countries committing to a reduction in this loss by 2020 (CBD, 2010). In order to achieve these commitments, there is an urgent need for financial investment and political leadership to help conserve biodiversity, but so far the response from the global community is insufficient. As such, the direction of scarce resources towards priority species and priority sites seems to be a key means towards achieving the Aichi targets. I have

explored some of these concepts in my thesis within a particular geographic region and a single taxonomic group. The issues that I have outlined for the Himalaya are typical of the choices that decision makers need to make across both the Himalaya and across the globe: in one sense, the Himalaya can be said to act as a microcosm of global conservation issues. In the following section, I discuss how some of the concepts I have explored can help achieve global biodiversity targets.

6.3.1 Applicability of my results to other taxa and locations

In order to assess how my thesis can help achieve global biodiversity targets, it is important to discuss whether the results I have found can be generalised to other taxa and other areas. There are a number of examples where this can be achieved: first, the species distribution modelling technique that I outline in Chapter 2 could be applied to other cryptic and endangered species, but requires further testing. A key requirement is that there are species with similar habitat requirements to the target species but the level of similarity that is required is not yet known. Second, it is possible to examine species declines in conservation status in the same way as I have shown in Chapter 3, using geographic range and correction factors to set priorities, but this depends on the amount of point locality data and whether population abundance is measured either directly or using a proxy. Third, the results I have found for Galliformes in Chapter 4 requires testing with other taxa. These results have bearings on other Galliformes species across the globe and potentially, for other taxa. Finally, the finding in Chapter 5 requires testing with other taxa. Specifically, there is a need to assess whether Galliformes' distributions are spatially congruent with other birds or groups, but our methods can be used with multiple taxonomic groups, other types of input data and across multiple environments (Girardello et al., 2009; Moilanen et al., 2008). While I found that Galliformes biodiversity within Himalaya could be better represented in the PA network, making recommendations for expansion may be less straightforward because local and global priorities often differ and my results may not be best way of achieving PA targets in a global sense (Woodley et al., 2012).

6.3.2 Spatial scale and conservation

Determining how to prioritise conservation action between species and sites over different geographic spatial scales is an important topic for investigation. There are many arguments both for and against prioritisation in a local setting (Pressey et al., 1993; Hunter and Hutchinson, 1994; Brown et al., 1995; Lesica and Allendorf, 1995; Kark et al., 1999; Channell and Lomolino, 2000).

For example, political decision-making units may not match well with species' distributions but as most conservation is parochial (Hunter and Hutchinson, 1994), it can be useful to target resources towards species in this fashion. For example, if a species is locally threatened but globally thriving should it be prioritised where it is threatened or where it is thriving? The answer surely depends in part on how much the focal area contributes towards the species' global range. My thesis provides an example of how this can be investigated in Chapter 3, which helps to show that geographical extent must be clearly defined in goal setting and conservation prioritisation, as priorities are often different at different geographic extents (Kark et al., 2009; Moilanen and Arponen, 2011). In this sense my thesis can help contribute towards global biodiversity targets, because examining such conservation problems at smaller spatial scales in a bottom-up fashion can help implement conservation action at the global scale (Mace et al., 2000; Whittaker et al., 2005; Brooks et al., 2006).

6.3.3 Data and decision-making: historical data

One other important bearing that this thesis has on our ability to achieve global biodiversity targets is in demonstrating the value of using historical data to address some fundamental, contemporary conservation questions. Conservation biology suffers from a paucity of data (e.g. Lomolino, 2004; Kozlowski, 2008; Cardoso et al., 2011) and it has been argued that if we hesitate to make conservation decisions in the face of incomplete data, we can delay management, which means biodiversity loss is likely to continue (e.g. Martin et al., 2012; Lindenmayer et al., 2013). At the same time, we need to better measure conservation impact and invest in successful strategies (Boakes et al., 2010). Pimm et al. (2014) demonstrated that there is a need for new data sources, which may include citizen science data, crowd sourcing (e.g. www.ebird.org, 2009) and also historical data. Although historical data has been posited as a partial solution to the lack of data in conservation, the question is: how to use this data to make conservation decisions when it contains many biases? In my thesis I provide a couple of examples of how to do this: in Chapter 2, I created extintion probabilities that incorporated measures of search effort; in Chapters 3 and 4, I used measures of species decline (both relative and absolute) that utilised correction factors that accounted for spatio-temporal biases in record collecting; in Chapters 2 and 5, I used SDMs that incorporated spatio-temporal record collection biases in Maxent software and used distribution discounting in Zonation software to account for uncertainty in my systematic conservation planning approach. Undertaking these kinds of analyses are vital for two main reasons: first, the problem of data shortages in conservation is likely to increase, especially given

the decisions to be made in the coming decades with pressure for anthropogenic land-use changes to increase rapidly; and second, birds are more well-known than any other taxonomic group (Larsen et al., 2012) and so data shortages are likely to be worse for other taxa.

One data issue that my thesis has highlighted is that museum specimen collection has declined as it is no longer considered morally acceptable and is not being replaced as a source of location records that can be repeatedly examined. Trip reports as collected through citizen science have been suggested as a solution (Pimm et al., 2014). However, in the thesis Appendix I show that some Galliform species are better represented through trip reports than others and the proportion of total Galliform records made up by trip reports in the Himalaya is very small, although it is increasing. Most trip reports come from Nepal and Bhutan and are locations that people return to regularly, which is similar to results shown by other studies (e.g. Tulloch et al., 2013). If trip reports were used to construct estimates of current species distributions within the Himalaya, then range sizes would be severely underestimated. Given that many of the other records come from specimen data, we may need to take a more active approach in directing citizen scientists to certain areas if trip reports are to help replace museum specimens (e.g. Schmeller et al., 2009).

Finally, although I have not explicitly examined this directly in my thesis, limited conservation data can also affect our ability to incorporate economic costs in a decision-making setting. For example, it has been stated that if we are to provide rational conservation priorities for planners and policy-makers, then the costs of conservation actions and the likelihood that conservation interventions will achieve the desired effect need to be considered (Naidoo et al., 2006; Wilson et al., 2009). This has been done using simple cost-effectiveness formulae (e.g. Joseph et al., 2009), but there are more sophisticated approaches available, such as the Marxan algorithm (Wilson et al., 2006). However, many of these approaches have not sufficiently accounted for data limitations, which could have a negative impact on our decision-making process (Armsworth, 2014). Advancements in this subject area could be extremely helpful both for the conservation of Galliformes in the Himalaya, but also for biodiversity in general. Overall, while it can be challenging to make decisions for conservation using limited data, it is necessary to do so and my thesis has illustrated a number of ways that it can be achieved.

6.4 Conclusion

In this thesis I have provided new insights into the conservation status and distributions of Himalayan Galliformes. Using a database of historical records and extinction probability calculations, I have shown that the Himalayan quail, which is the rarest Galliform species that is found in the Himalaya, may still be extant. In order to help test this finding, I have provided an updated map of its potential distribution using a novel technique that uses similar habitat requirements of other more common species. This is directing field scientists (through the RSPB and BirdLife Partnership) and an array of highly skilled amateur birdwatchers to sites that are most likely to have suitable habitat for this species. Furthermore, I have examined long-term species declines in conservation status for Himalayan Galliformes both within and outside of the Himalaya and have used those measures in conjunction with endemism to the Himalaya to identify regional conservation priorities. Two species which are considered as 'Least Concern' by the IUCN Red List were identified as requiring a review of their regional conservation status. The methods I have used provides a means of assessing conservation status using non-systematically collected data across a variety of spatial scales over long time periods. More broadly, I have assessed whether long-term declines in geographically widespread species have been at a faster rate than those in restricted-range species and whether these declines are likely or not to accord with contemporary conservation status. I found that declines in widespread species are unlikely to be different in the rate of decline to more restricted species and there is no general pattern with IUCN Red List status. This suggests that in the Galliformes bird order there is no urgent need to develop different techniques for assessing the conservation status in widespread species. Finally, I have created new SDMs for Himalayan Galliformes and assessed how well the current PA network represents species of conservation concern as measured across different values. I found that the network fails to represent endemic and threatened species and could be re-designed to better account for these Galliform species. These findings will help to conserve a highly threatened avian order within a region identified as being important for conservation and help provide information towards a region-wide Species Conservation Strategy. They provide some important examples of how resources can be directed to priority species and priority sites within the Himalaya. Undertaking such analyses and treating Galliformes in a regional sense are invaluable, because people care about what they know and what is close to their daily lives. Similarly, I provide examples of how we can make use of historical data, which is sometimes dismissed due to data biases. Undertaking these priority-setting exercises across a variety of scales and using

whatever data is available to us is extremely important if we are to scale-up such exercises to the global scale and make efficacious conservation responses. This is likely to help play an important part in tackling global biodiversity loss.

Appendix

Delimiting the Himalaya

Incorporating the Himalaya into a spatially explicit study requires that the area is unequivocally delimited and outlined. The delimitation of my study area both reflects natural patterns and political units. In other studies that have different aims and imperatives, the Greater Himalaya has been defined to include general areas of high mountains and plateaus in Central, South and Inner Asia (Xu et al., 2009). However, for this project Greater Himalaya was defined using 11 WWF ecoregions (see Figure A.1) as a spatial boundary. This area intersects five countries, the details of which are shown in Table A.1.

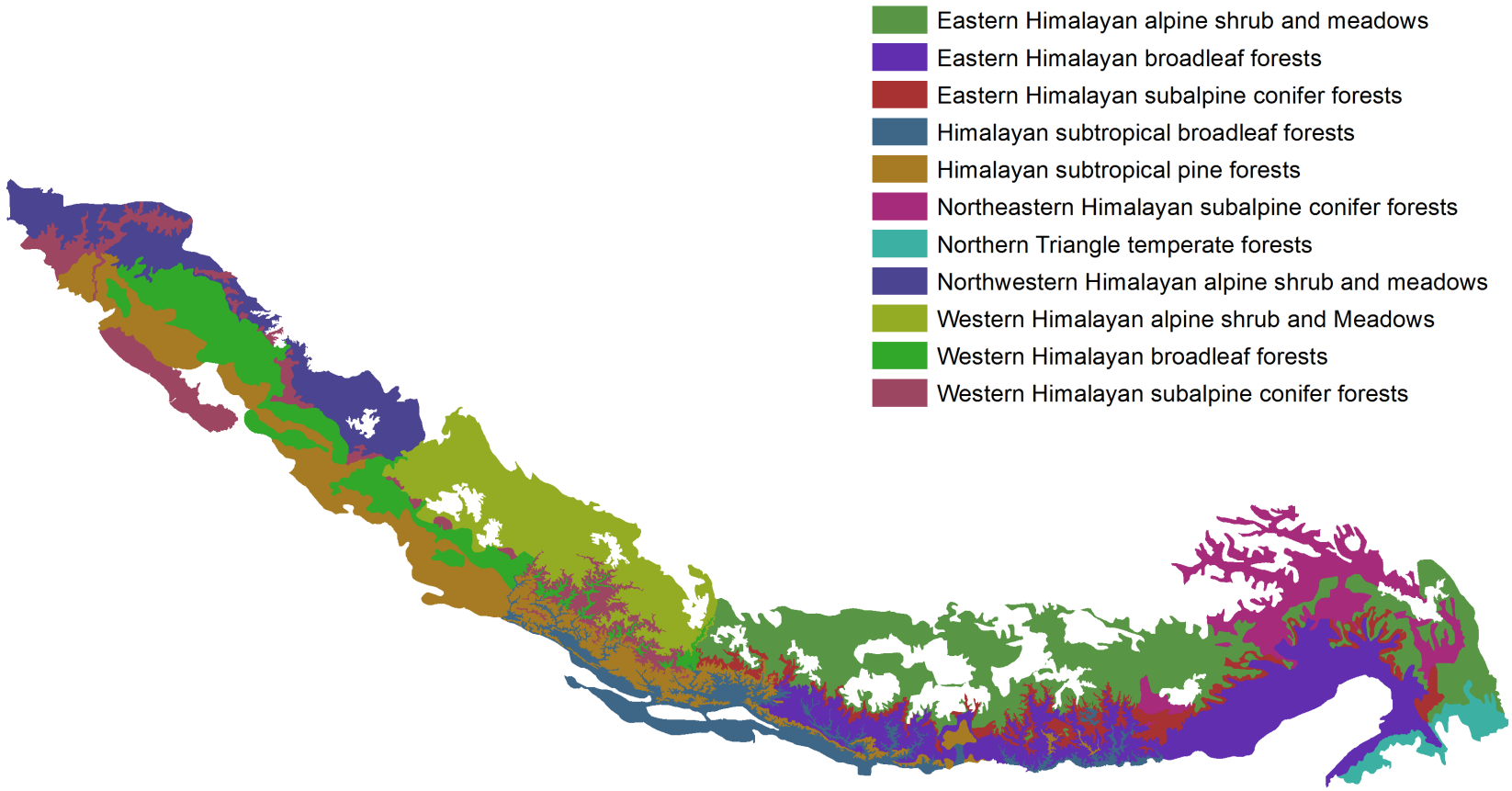


Figure A.1: Map of WWF ecoregions used in thesis to delimit Himalayan study region.

The term 'ecoregion' is used by WWF to define biogeographic units that represent all habitats and distinct biotas in networks of conservation areas (Olson et al., 2001). They were originally proposed to identify areas with outstanding levels of biodiversity and to represent different natural communities. The boundaries approximate to the approximate extent of natural communities prior to major land-use changes (Olson et al., 2001). WWF's ecoregions have a dual function of reflecting biogeographic patterns and acting as tractable units for conservation action at both global and regional scales (Olson et al., 2001). While there have been criticisms from biogeographers (e.g. Jepson and Whittaker, 2002), most conservation NGOs have adopted WWF ecoregions in their planning frameworks as the appropriate spatial link between global priority-setting and site-based conservation action (Wikramanayake et al., 2002a).

Name	Area within study site km^2	Proportion total area of study site	Total area km^2	Proportion total area within study site	Number of PAs within study site	Proportion of PA network within study site	Number of Galliform species within study site	GDP millions of US$ (IMF 2013)
Bhutan	37625	0.06	38394	0.98	9	0.07	14	1985
China	184232	0.28	9326410	0.02	14	0.46	21	9181377
India	226372	0.35	2864021	0.08	119	0.24	23	1870651
Myanmar	16192	0.02	653508	0.02	5	0.06	10	56408
Nepal	127478	0.20	143351	0.89	31	0.15	16	19341
Pakistan	60880	0.09	856692	0.07	29	0.01	11	238737

Table A.1: Further information on the 6 countries within the study region. GDP estimates taken from International Monetary Fund (2013).

Himalayan Galliformes

The following section provides images of each galliform species (Figure A.2), information on the conservation status and threats across all galliform species in their Himalaya (Figure A.3) and also a breakdown of the aforementioned data for each species (Tables A.2, A.3 and A.4).

Figure A.2: Photographs of resident Himalayan galliform species. All pictures are from Arkive (www.arkive.org, 2014) or the World Pheasant Association. Species include: A = blood pheasant (*Ithaginis crutentus*), B = blyth's tragopan (*Tragopan blythii*), C = buff-throated partridge (*Tetraophasis szechenyii*), D = cheer pheasant (*Catreus wallichi*), E = chestnut-breasted partridge (*Arborophila mandelli*), F = chukar (*Alectoris chukar*), G = common quail (*Coturnix coturnix*), H = hill partridge (*Arborophila torqueola*), I = Himalayan monal (*Lophophorus impejanus*), J = Himalayan quail (*Ophrysia superciliosa*), K = Himalayan snowcock (*Tetraogallus himalayensis*), L = Indian peafowl (*Pavo cristatus*), M = kalij pheasant (*Lophura leucomelanos*), N = koklass pheasant (*Pucrasia macrolopha*), O = red junglefowl (*Gallus gallus*), P = rufous-throated partridge (*Arborophila rufogularis*), Q = satyr tragopan (*Tragopan satyra*), R = sclater's monal (*Lophophorus sclateri*), S = snow partridge(*Lerwa lerwa*), T = temminck's tragopan (*Tragopan temminckii*), U = Tibetan-eared pheasant (*Crossoptilon harmani*), V = Tibetan partridge (*Perdix hodgsoniae*), W = Tibetan snowcock (*Tetraogallus tibetanus*), X = western tragopan (*Tragopan melanocephalus*). N = 24 species.

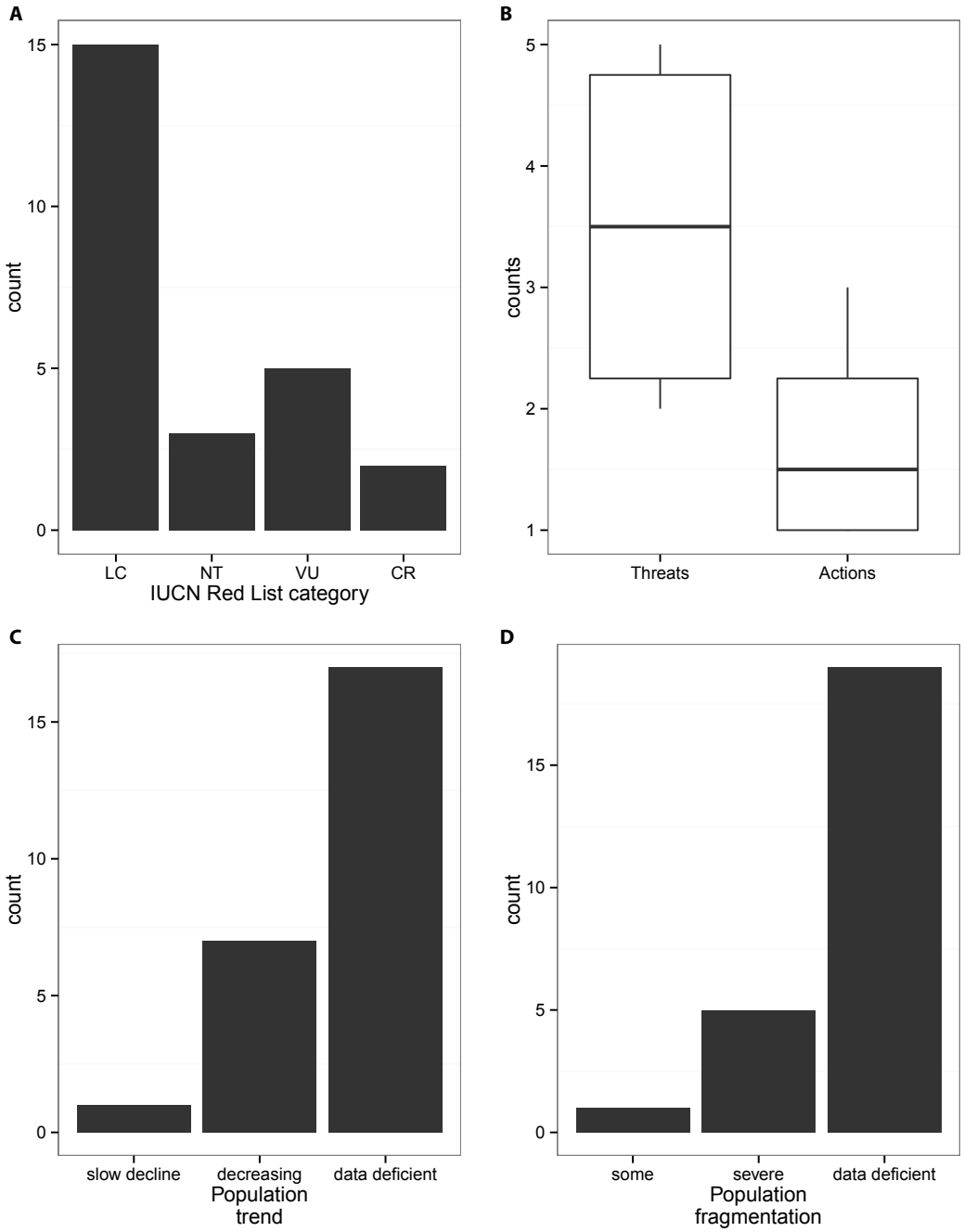


Figure A.3: Conservation status graphs. A) Bar graph showing the number of Himalayan galliform species by their global extintion status; B) box and whisker plot showing the median number of threats and conservation actions across all Himalayan galliform species; C) bar graph showing the number of Himalayan galliform species by their global population trends; D) bar graph showing the number of Himalayan galliform species by their global population fragmentation

Common name	Binomial	Sub-species (Y/N)	IUCN Red List category	Total number of countries	Total number of Greater Himalayan countries	Number of habitats	Alpine obligates (Y/N)
Blood pheasant	*Ithaginis cruentus*	Y	LC	5	5	2	Y
Blyth's tragopan	*Tragopan blythii*	N	VU	4	4	DD	N
Buff-throated partridge	*Tetraophasis szechenyii*	N	LC	2	2	4	N
Cheer pheasant	*Catreus wallichi*	Y	VU	3	3	2	N
Chestnut-breasted partridge	*Arborophila mandelli*	N	VU	3	3	2	N
Chukar	*Alectoris chukar*	N	LC	45	5	3	N
Common quail	*Coturnix coturnix*	N	LC	127	6	4	N
Hill partridge	*Arborophila torqueola*	N	LC	8	5	3	N

Table A.2: continued on next page

Common name	Binomial	Sub-species (Y/N)	IUCN Red List category	Total number of countries	Total number of Greater Himalayan countries	Number of habitats	Alpine obligates (Y/N)
Himalayan monal	*Lophophorus impejanus*	Y	LC	7	7	3	Y
Himalayan quail	*Ophrysia superciliosa*	N	CR	1	1	2	N
Himalayan snowcock	*Tetraogallus himalayensis*	N	LC	11	5	2	N
Indian peafowl	*Pavo cristatus*	N	LC	9	4	5	N
Kalij pheasant	*Lophura leucomelanos*	N	LC	8	6	3	N
Koklass pheasant	*Pucrasia macrolopha*	Y	LC	5	5	1	N
Red junglefowl	*Gallus gallus*	N	LC	27	6	4	N
Rufous-throated partridge	*Arborophila rufogularis*	N	LC	9	5	3	N
Satyr tragopan	*Tragopan satyra*	Y	NT	4	4	1	N

Table A.2: continued on next page

Common name	Binomial	Sub-species (Y/N)	IUCN Red List category	Total number of countries	Total number of Greater Himalayan countries	Number of habitats	Alpine obligates (Y/N)
Sclater's monal	*Lophophorus sclateri*	N	VU	3	3	3	Y
Snow partridge	*Lerwa lerwa*	N	LC	6	6	3	N
Temminck's tragopan	*Tragopan temminckii*	N	LC	4	3	2	N
Tibetan-eared pheasant	*Crossoptilon harmani*	N	NT	2	2	2	Y
Tibetan partridge	*Perdix hodgsoniae*	N	LC	4	4	3	N
Tibetan snowcock	*Tetraogallus tibetanus*	N	LC	5	4	2	N
Western tragopan	*Tragopan melanocephalus*	Y	VU	2	2	1	N

Table A.2: The conservation status for each Himalayan galliform species part 1. Key: LC = 'least concern', NT = 'near threatened', VU = 'vulnerable', EN = 'endangered' and CR = 'critically endangered'. N = 24 species.

Common name	Binomial	Endemic to Himalaya (Y/N)	EOO km^2	Alt. range m	Alt. migrant (Y/M)	Est. pop.	Number sub-pops.	Fragment'n	Pop. trend
Blood pheasant	*Ithaginis cruentus*	DD	DD	2,500 / 4,500	N	DD	DD	DD	DD
Blyth's tragopan	*Tragopan blythii*	DD	42,400	1,400-3,300	Y	2,500 / 9,999	>1	severe	dec.
Buff-throated partridge	*Teraophasis szechenyii*	DD	DD	3,300 / 4,600	N	DD	DD	DD	DD
Cheer pheasant	*Catreus wallichi*	W Himalaya	27,500	1,200 / 3,250	N	4,000 / 6,000	>1	severe	dec.
Chestnut-breasted partridge	*Arborophilia mandellii*	DD	67,100	350 / 2,500	N	2,500 / 9,999	>1	severe	dec.
Chukar	*Alectoris chukar*	N	10,300,000	0 / 4,500	N	2,000,000-10,000,000	DD	DD	DD
Common quail	*Cortunix cortunix*	N	DD	0 / 1000	N	35 / 300,000	DD	DD	DD

Table A.3: continued on next page

Common name	Binomial	Endemic to Himalaya (Y/N)	EOO km^2	Alt. range m	Alt. migrant (Y/M)	Est. pop.	Number sub-pops.	Fragment'n	Pop. trend
Hill partridge	*Arborophilia torqueola*	N	DD	1,500 / 2,700	N	DD	DD	DD	DD
Himalayan monal	*Lophophorus impejanus*	Both	481,000	2,100 / 4,500	Y	DD	DD	DD	DD
Himalayan quail	*Ophyrsia superciliosa*	Y	DD	1,650 / 2,400	N	<50	DD	DD	DD
Himalayan snowcock	*Tetraogallus himalayensis*	N	1,910,000	3,600 / 4570	N	200,000	DD	DD	DD
Indian peafowl	*Pavo cristatus*	N	2,840,000	0 / 2,000	N	DD	DD	DD	DD
Kalij pheasant	*Lophura leumcomelanos*	Both	DD	DD	N	DD	DD	DD	DD
Koklass pheasant	*Pucrasia macrolopha*	W Himalaya	DD	2,000 / 4,000	Y	DD	DD	DD	DD
Red junglefowl	*Gallus gallus*	Both	DD	0 / 2,000	N	DD	DD	DD	DD

Table A.3: continued on next page

Common name	Binomial	Endemic to Himalaya (Y/N)	EOO km^2	Alt. range m	Alt. migrant (Y/M)	Est. pop.	Number sub-pops.	Fragment'n	Pop. trend
Rufous-throated partridge	*Arborophilia rufogularis*	N	DD	300 / 2,600	N	DD	DD	DD	DD
Satyr tragopan	*Tragopan satyra*	Both	99,200	2,100 / 4,250	Y	10,000 / 19,999	DD	DD	dec.
Sclater's monal	*Lophophorus sclateri*	Y	45,500	2,000 / 4,200	N	2,500 / 9,999	>1	severe	dec.
Snow partridge	*Lerwa lerwa*	DD	DD	2,500 / 5,000	N	100,000 / 499,99	DD	DD	DD
Temminck's tragopan	*Tragopan temminckii*	DD	1,260,000	DD	Y	100,000 / 499,999	DD	DD	DD
Tibetan eared pheasant	*Crossoptilon harmani*	DD	DD	DD	N	DD	DD	DD	dec.
Tibetan partridge	*Perdix hodgsoniae*	DD	DD	3,600 / 5,600	N	DD	DD	DD	DD
Tibetan snowcock	*Tetraogallus thibetanus*	DD	DD	3,700 / 5,800	N	1,00,000 / 499,999	DD	DD	DD

Table A.3: continued on next page

Common name	Binomial	Endemic to Himalaya (Y/N)	EOO km^2	Alt. range m	Alt. migrant (Y/M)	Est. pop.	Number sub-pops.	Fragment'n	Pop. trend
Western tragopan	*Tragopan melanocephalus*	W Himalaya	21,600	1,750 / 3,600	Y	5,000	>1	severe	slow decline

Table A.3: The conservation status for each Himalayan galliform species part 2. Key: EOO = extent of occurrence, alt. = altitudinal, pop. = population, DD = data deficient, dec. = declining. N = 24 species.

Common name	Binomial	Number of threats	Number of conservation actions underway	Conservation actions proposed (Y/M)
Blood pheasant	*Ithaginis cruentus*	DD	DD	DD
Blyth's tragopan	*Tragopan blythii*	4	3	Y
Buff-throated partridge	*Teraophasis szechenyii*	DD	DD	DD
Cheer pheasant	*Catreus wallichi*	5	3	Y
Chestnut-breasted partridge	*Arborophilia mandellii*	5	1	Y
Chukar	*Alectoris chukar*	DD	DD	DD
Common quail	*Cortunix cortunix*	DD	DD	DD
Hill partridge	*Arborophilia torqueola*	DD	DD	DD
Himalayan monal	*Lophophorus impejanus*	DD	DD	DD
Himalayan quail	*Ophyrsia superciliosa*	3	DD	Y
Himalayan snowcock	*Tetraogallus himalayensis*	DD	DD	DD
Indian peafowl	*Pavo cristatus*	DD	DD	DD
Kalij pheasant	*Lophura leumcomelanos*	DD	DD	DD
Koklass pheasant	*Pucrasia macrolopha*	DD	DD	DD
Red junglefowl	*Gallus gallus*	2	DD	DD
Rufous-throated partridge	*Arborophilia rufogularis*	DD	DD	DD

Table A.4: continued on next page

Common name	Binomial	Number of threats	Number of conservation actions underway	Conservation actions proposed (Y/M)
Satyr tragopan	*Tragopan satyra*	4	2	DD
Sclater's monal	*Lophophorus sclateri*	2	1	Y
Snow partridge	*Lerwa lerwa*	DD	DD	DD
Temminck's tragopan	*Tragopan temminckii*	DD	DD	DD
Tibetan eared pheasant	*Crossoptilon harmani*	2	1	DD
Tibetan partridge	*Perdix hodgsoniae*	DD	DD	DD
Tibetan snowcock	*Tetraogallus thibetanus*	DD	DD	DD
Western tragopan	*Tragopan melanocephalus*	5	2	Y

Table A.4: Breakdown in the number of pressures and conservation actions for each Himalayan galliform species. Key: DD = data deficient. N = 24 species.

Data biases

Data collection has been uneven throughout space (see Figure A.4) and time (see Figure A.5). Similarly, the proportions of each species' global range contained within the Himalaya differed markedly (see Figure A.6). We excluded the following records: 1) if the record was obviously outside of the natural range of the species (i.e. records of Indian peafowl in Spain); 2) if the records did not have latitude/longitude co-ordinates or a collection date. The effect of this exclusion procedure on the number of records available for analysis can be seen in Figure A.7.

More records have been collected throughout time (see Figure A.8) but also in different ways. For example, museum record collection has declined but the number of reference records has risen (see Figure A.9). This has affected the types of observation record, but also the geo-referencing accuracy of such records (see Figure A.9).

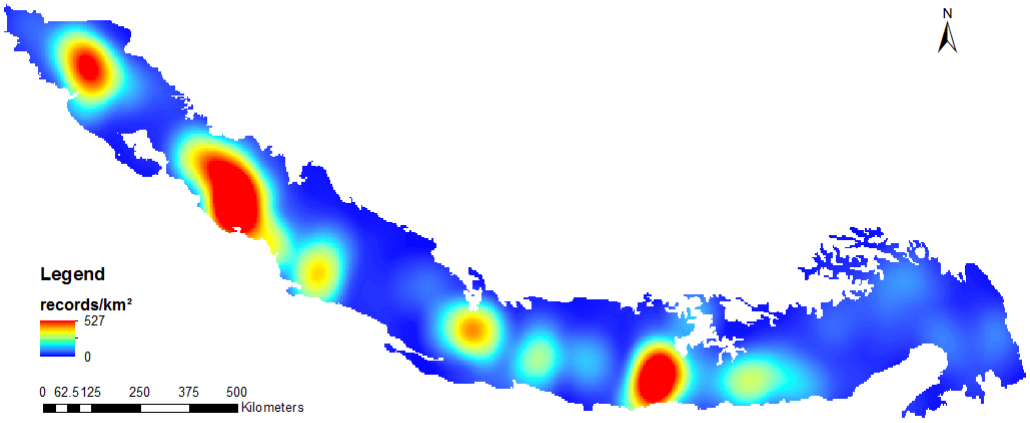


Figure A.4: Kernel density map showing the uneven distribution of record collection throughout the Himalaya across all time periods. Warm colours indicate areas with high numbers of records per km^2 and cool colours indicate areas with low numbers of records per km^2.

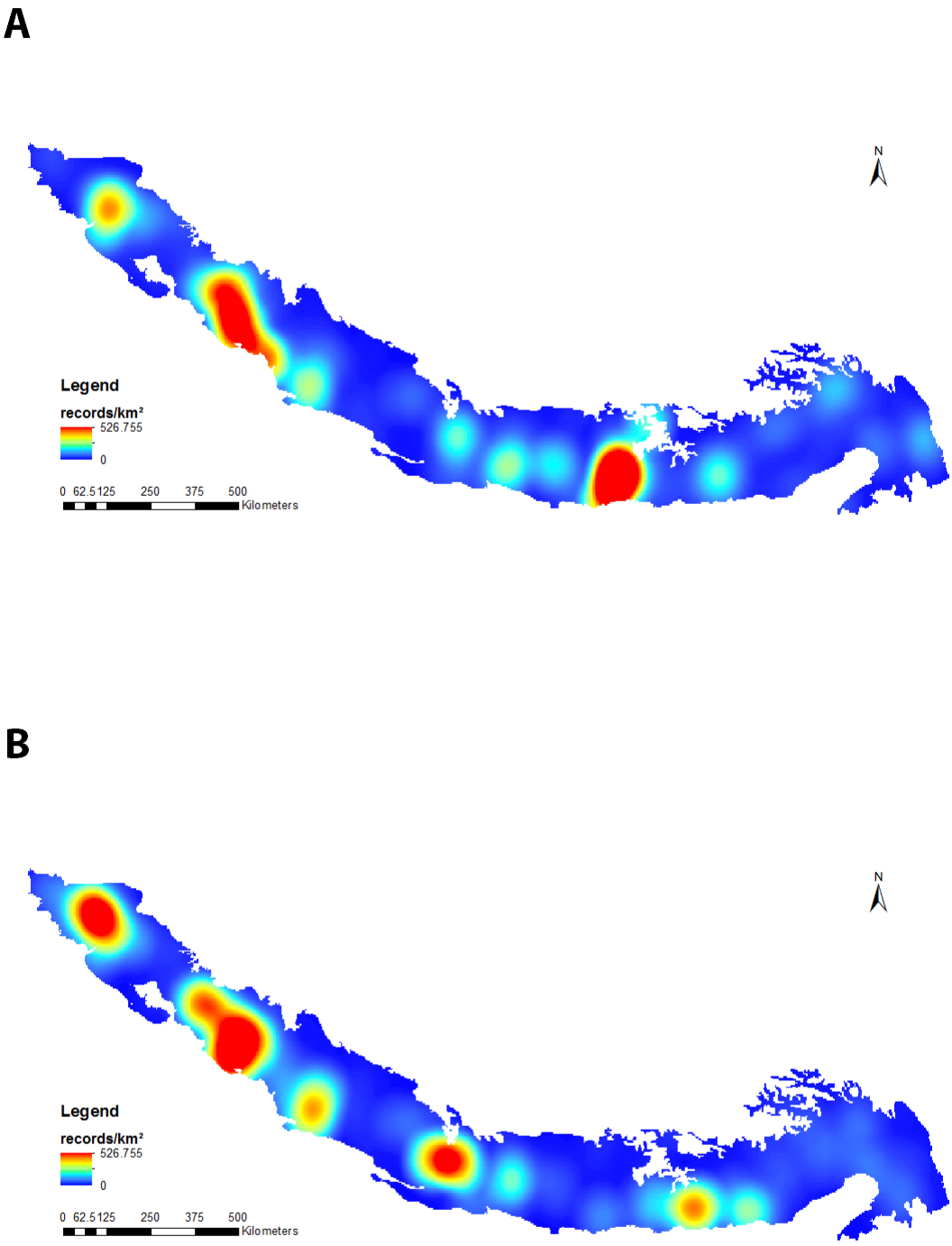


Figure A.5: Kernel density maps showing the uneven distribution of record collection throughout the Himalaya across A) pre-1980 and B) post-1980. Warm colours indicate areas with high numbers of records per km^2 and cool colours indicate areas with low numbers of records per km^2.

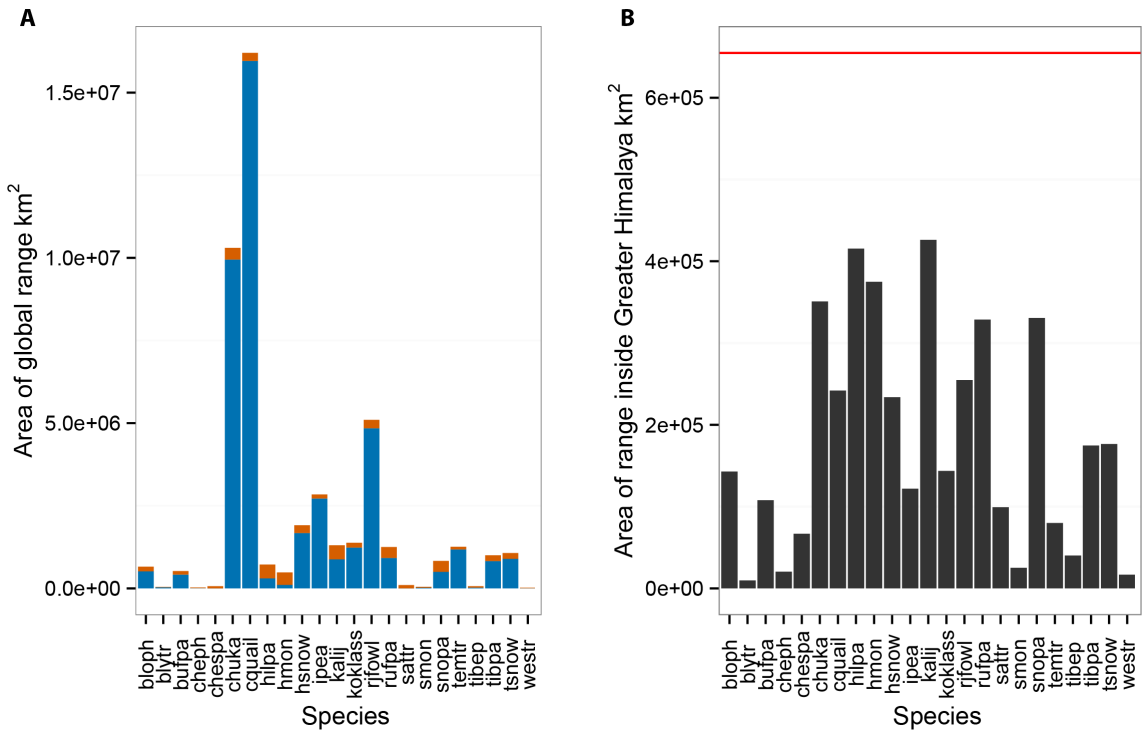


Figure A.6: Bar graphs of Galliform ranges. A) shows the size of each Himalayan Galliformes species global geographic range and the respective proportions of the range inside (coloured orange) and outside (coloured blue) of the Greater Himalaya. Range size estimates were taken from BirdLife and reflect Extent of Suitable Habitat. B) shows the area of range inside the Greater Himalaya for each Galliform species. The red horizontal line represents the total area of the Himalaya (654,772 km^2). Thus species with a range area closer to the red line have ranges that fill more of the Himalaya. Key: bloph = blood pheasant, blytr = Blyth's tragopan, bufpa = buff-throated partridge, cheph = cheer pheasant, chespa = chestnut-breasted partridge, chuka = chukar, cquail = common quail, hilpa = hill partridge, hmon = Himalayan monal, ipea = Indian peafowl, kalij = kalij pheasant, koklass = koklass pheasant, rjfowl = red junglefowl, rufpa = rufous-throated partridge, sattr = satyr tragopan, smon = sclater's monal, snopa = snow partridge, temtra = Temminck's tragopan, tibep = Tibetan eared pheasant, tibpa = Tibetan partridge, tsnow = Tibetan snowcock, westr = western tragopan. N = 23 species (due to data paucity, the Himalayan quail range is not shown here).

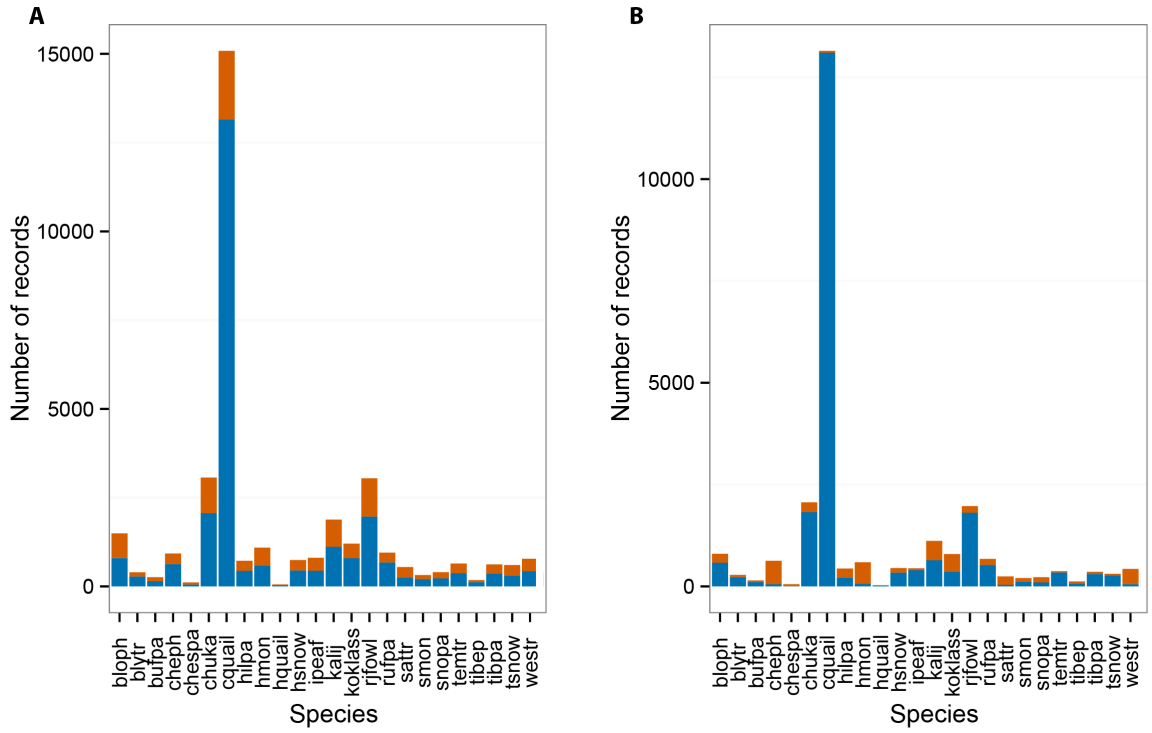


Figure A.7: Bar graphs of useable vs. non-useable Galliform records. A) shows the number of useable (blue) vs. non-useable (orange) records across the globe for Himalayan Galliformes. B) shows the number of useable records inside (orange) the Greater Himalaya vs. outside (blue) the Greater Himalaya for Himalayan Galliformes. Key: bloph = blood pheasant, blytr = Blyth's tragopan, bufpa = buff-throated partridge, cheph = cheer pheasant, chespa = chestnut-breasted partridge, chuka = chukar, cquail = common quail, hilpa = hill partridge, hmon = Himalayan monal, ipea = Indian peafowl, kalij = kalij pheasant, koklass = koklass pheasant, rjfowl = red junglefowl, rufpa = rufous-throated partridge, sattr = satyr tragopan, smon = sclater's monal, snopa = snow partridge, temtra = Temminck's tragopan, tibep = Tibetan eared pheasant, tibpa = Tibetan partridge, tsnow = Tibetan snowcock, westr = western tragopan. N = 23 species (Himalayan quail is not shown here as it merits treatment on its own; see Chapter 2)

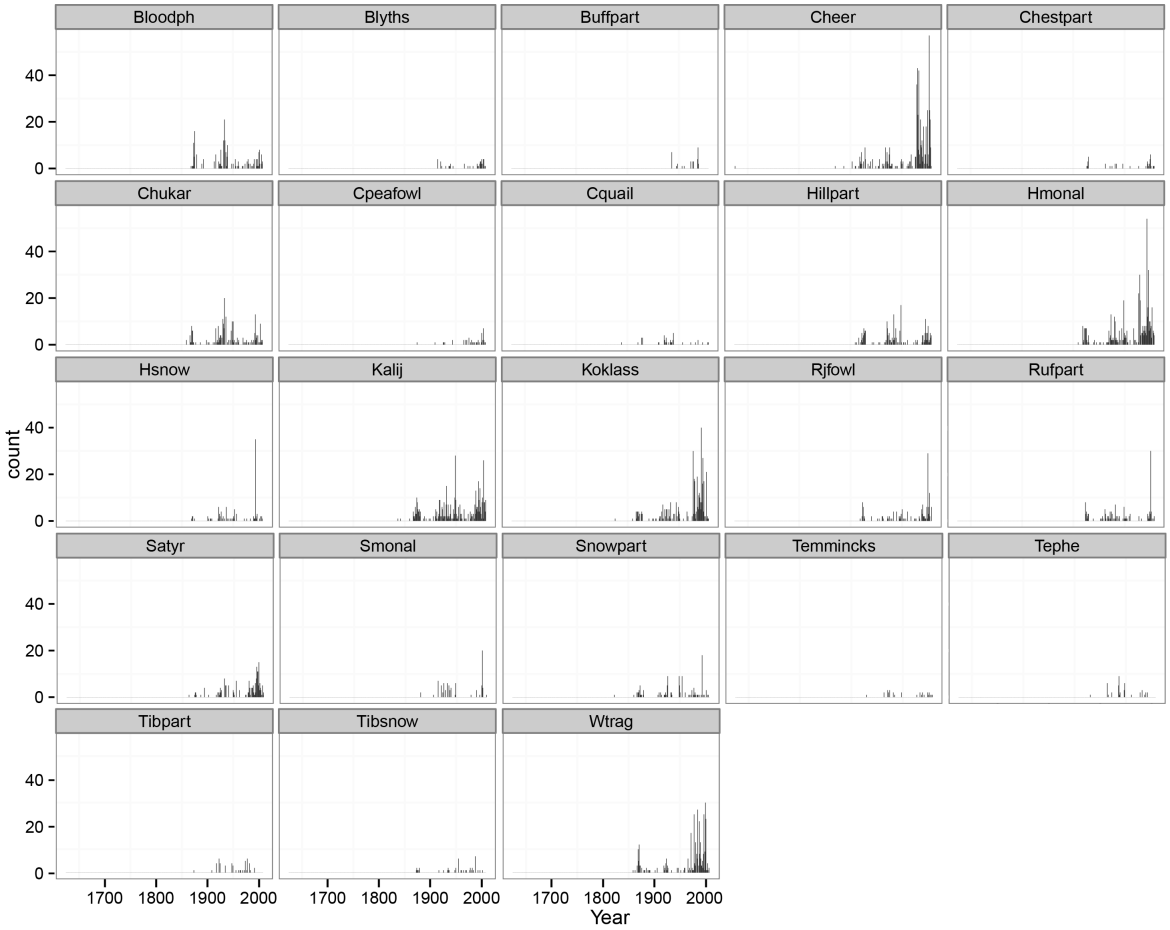


Figure A.8: Histograms showing the number of records for each Himalayan galliform species within the Himalaya throughout time. There is a general tendency for the number of records to increase with time. Key: Bloodph = blood pheasant, Blyths = Blyth's tragopan, Buffpart = buff-throated partridge, Cheer = cheer pheasant, Chestpart = chestnut-breasted partridge, Chukar = chukar, cpeafowl = common peafowl, Cquail = common quail, Hillpart = Hill partridge, Hmonal = Himalayan monal, Hsnow = Himalayan snowcock, Kalij = kalij pheasant, Koklass = koklass pheasant, Rjfowl = red junglefowl, Rufpart = rufous-throated partridge, Satyr = satyr tragopan, Smonal = Sclater's monal, Snowpart = snow partridge, Temmincks = Temminck's tragopan, Tephe = Tibetan-eared pheasant, Tibpart = Tibetan partridge, Tibsnow = Tibetan snowcock, Wtrag = western tragopan. N = 23 species (Himalayan quail is treated separately in Chapter 2).

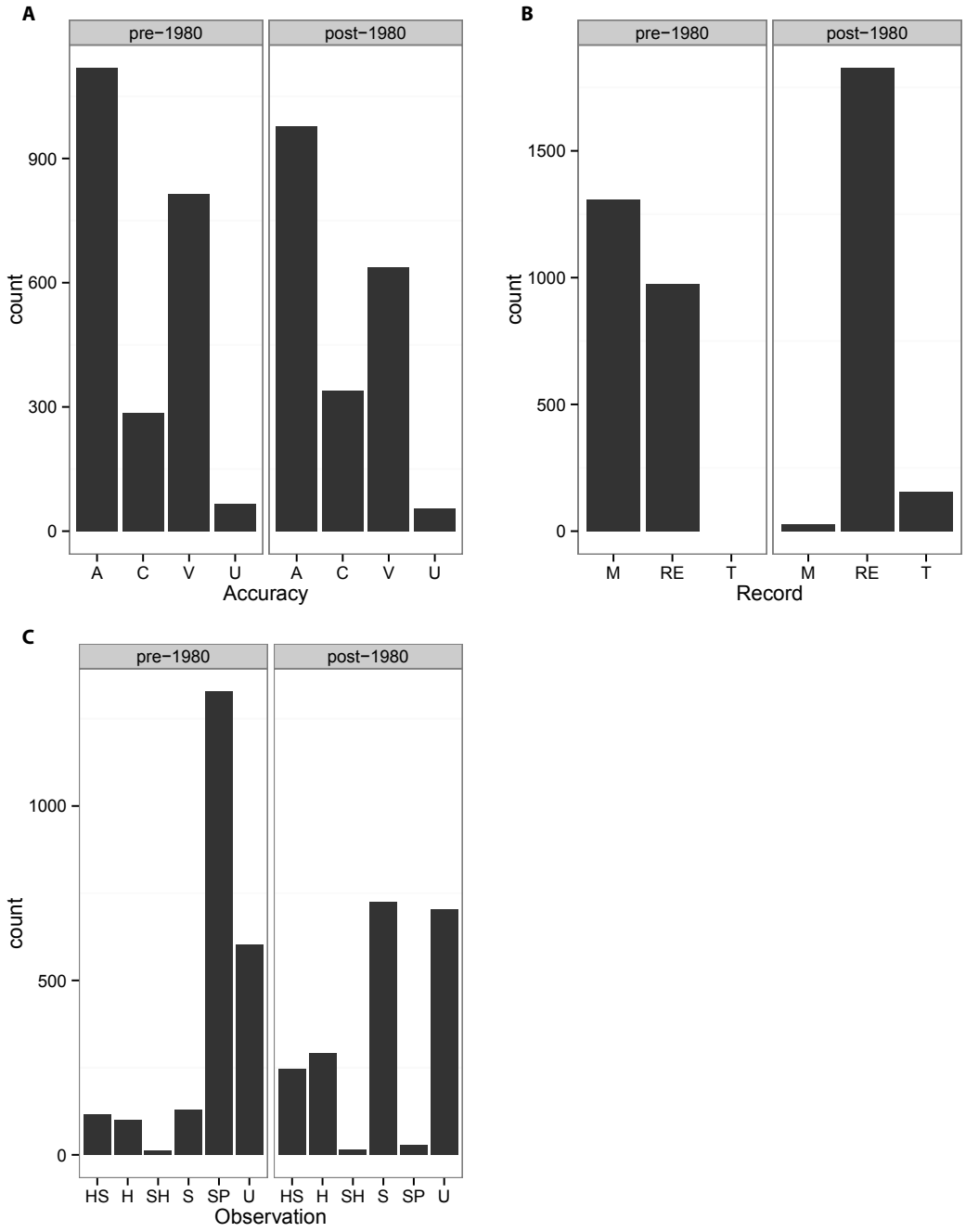


Figure A.9: Bar graphs of Galliform records through time. A) shows the accuracy of Himalayan Galliformes records pre- and post-1980. A = Accurate, C = Close, V = Vague, U = Unknown. B) shows the record type of Himalayan Galliformes records pre- and post-1980. M = Museum, RE = Reference, T = Trip report. C) shows the observation type of Himalayan Galliformes records pre- and post-1980. HS = Heard and Seen, H = Heard record, SH = Second Hand, S = Sight record, SP = Specimen, U = Unknown. N = 4293 records from 23 species (Himalayan quail is treated separately in Chapter 2) collected between 1625-2007 (383 years).

Trip reports

Museum specimens have been declining as it becomes a less acceptable technique of documenting biodiversity (Figure A.10). The challenge is how to track species' statuses and distributions - a possible replacement could be trip reports, which are more common as the increasingly affluent bird watcher community tends to go to more exotic places. So far, in the Himalaya trip reports are not evenly distributed between species and some species receive fewer sightings (χ^2 = 353.99, n = 157, df = 24, p<0.001), in part due to rareness (Figure A.11). However, it is clear that trip reports are focused in similar (non-random) areas in the Himalaya (χ^2 = 168.17, df = 38, p<0.001; Figure A.12). This pattern also holds true across time, with the number of return visits to each site non-random (χ^2 = 60.54, n = 39, df = 5, p<0.001; Figure A.13). Nepal and Bhutan have the greatest number of trip reports and are where people keep returning. As a result, if we are to be increasingly reliant on trip reports as a way of constructing species' distribution maps, we could be seriously underestimating their total ranges (compare Figure A.14 to Figure A.15 for range size estimates of the Himalayan monal). For example, if we construct AOO maps for a subset of 5 species, we find that the AOO value are significantly smaller if trip-reports alone are used to contruct our distribution maps in comparison to all other types of records (Figure A.16).

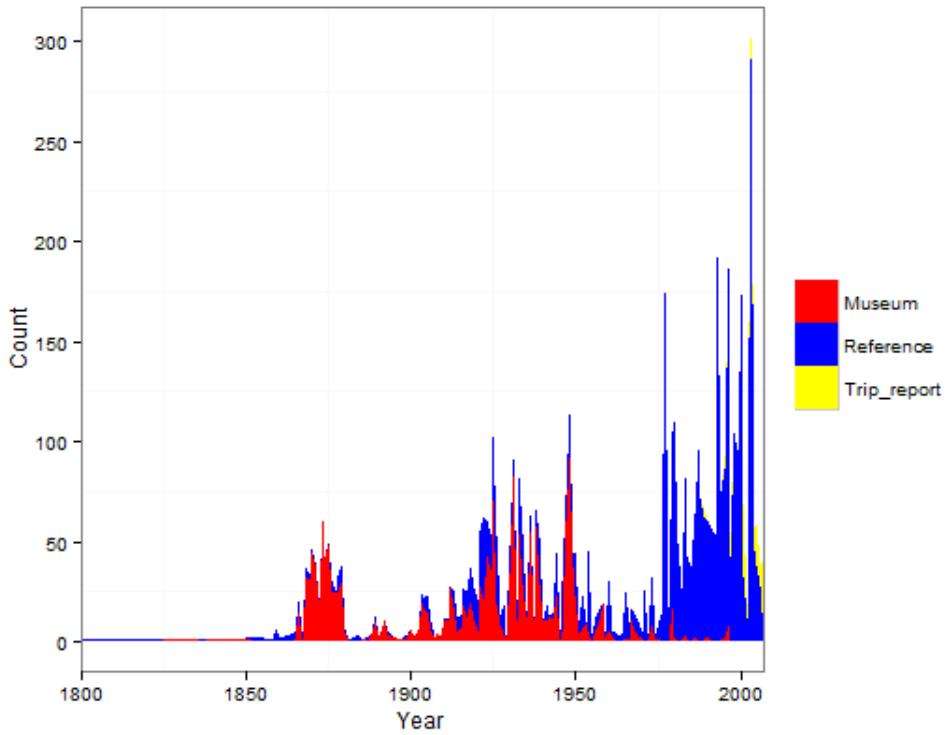


Figure A.10: Histogram showing the number of trip reports through time. Note: trip reports increase proportionally from the year 2000.

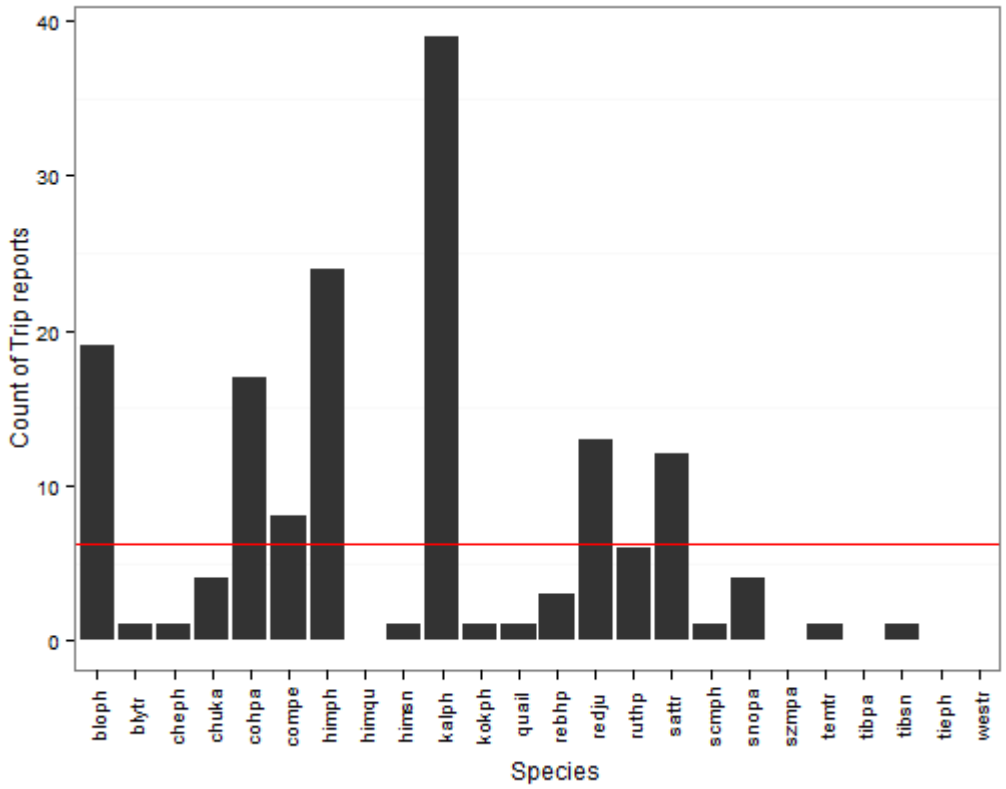


Figure A.11: Bar graph showing the number of trip reports for each Himalayan Galliform species. The red bar represents the median number (8). Blood pheasant, hill partridge, himalayan monal, kalij pheasant, red junglefowl and satyr tragopan have the most trip report counts. Key: bloph = blood pheasant, blytr = Blyth's tragopan, cheph = cheer pheasant, chuka = chukar, cohpa = hill partridge, compe = common peafowl, himph = Himalayan monal, himqu = Himalayan quail, himsn = Himalayan snowcock, kalph = kalij pheasant, kokph = koklass pheasant, quail = common quail, rebhp = red-breasted hill partridge, redju = red junglefowl, ruthp = rufous-throated partridge, sattr = satyr tragopan, scmph = sclater's monal, snopa = snow partridge, szmpa = buff-throated partridge, temtr = Temminck's tragopan, tibpa = Tibetan partridge, tibsn = Tibetan snowcock, tieph = Tibetan-eared pheasant, westr = western tragopan. N = 23 species.

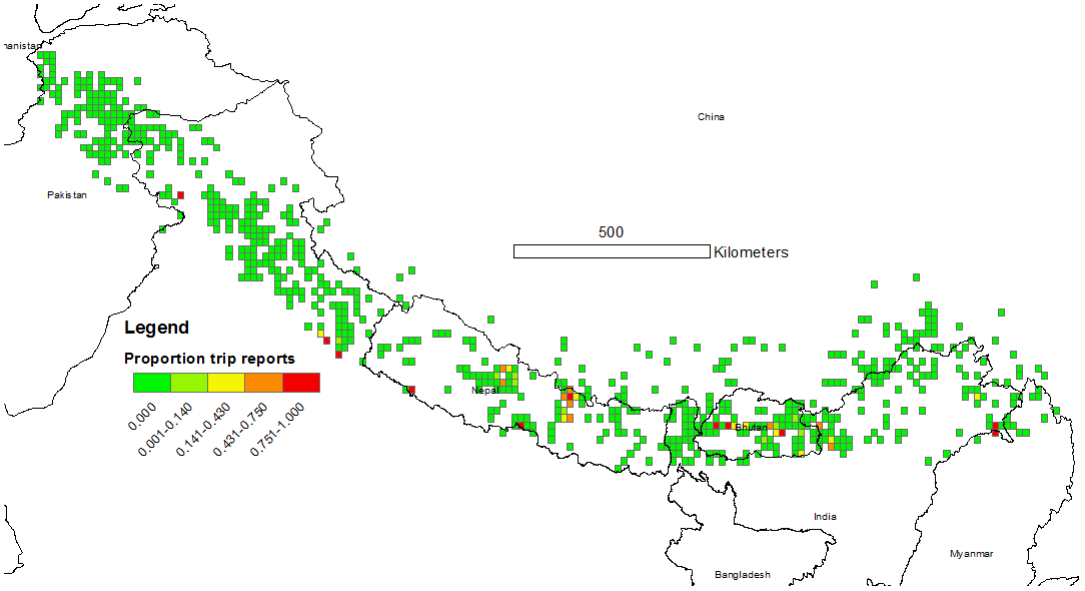


Figure A.12: Map showing the proportion of point locality records made up by trip reports in space. Grid squares are approximately 20 km^2 in size.

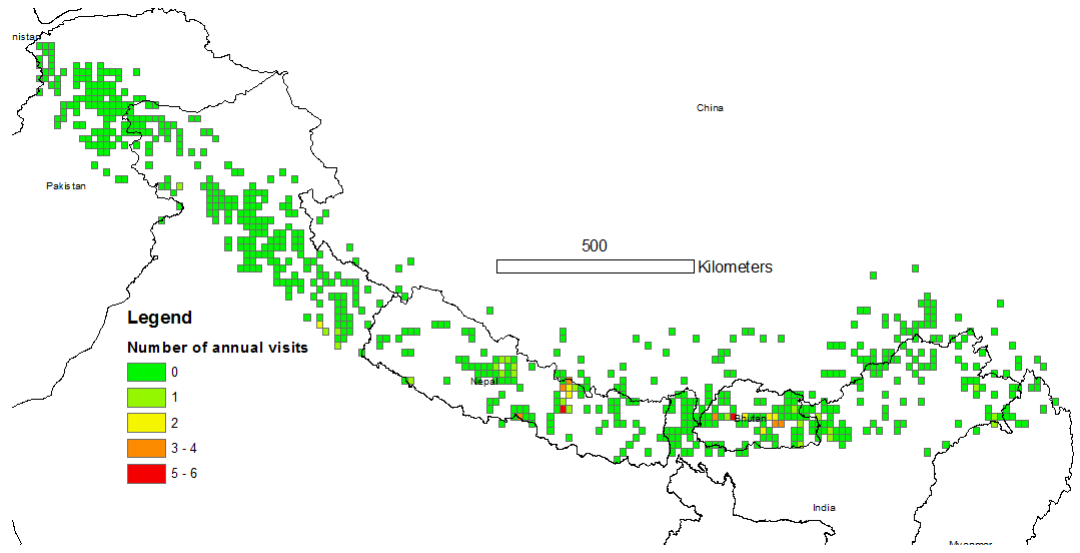


Figure A.13: Map showing the number of annual visits made to each locality. Grid squares are approximately 20 km^2 in size. Moran's i statistic revealed a significant clustering of visits (i = 0.16, z = +14.04, p<0.001).

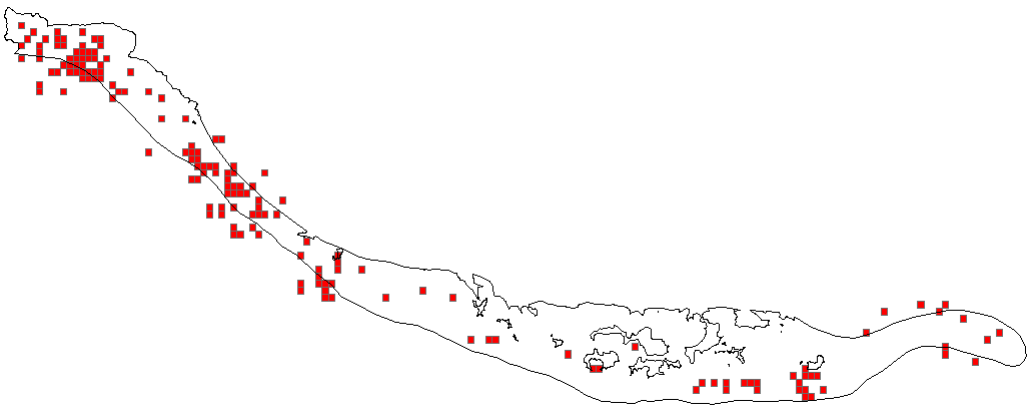


Figure A.14: Himalayan monal range constructed with all types of records but excluding trip reports. Grid squares are approximately 20 km^2 in size. Coloured squares represent area of occurrence and the polygon represents the current BirdLife range map.

Figure A.15: Himalayan monal range constructed with trip reports only. Grid squares are approximately 20 km^2 in size. Coloured squares represent area of occurrence and the polygon represents the current BirdLife range map.

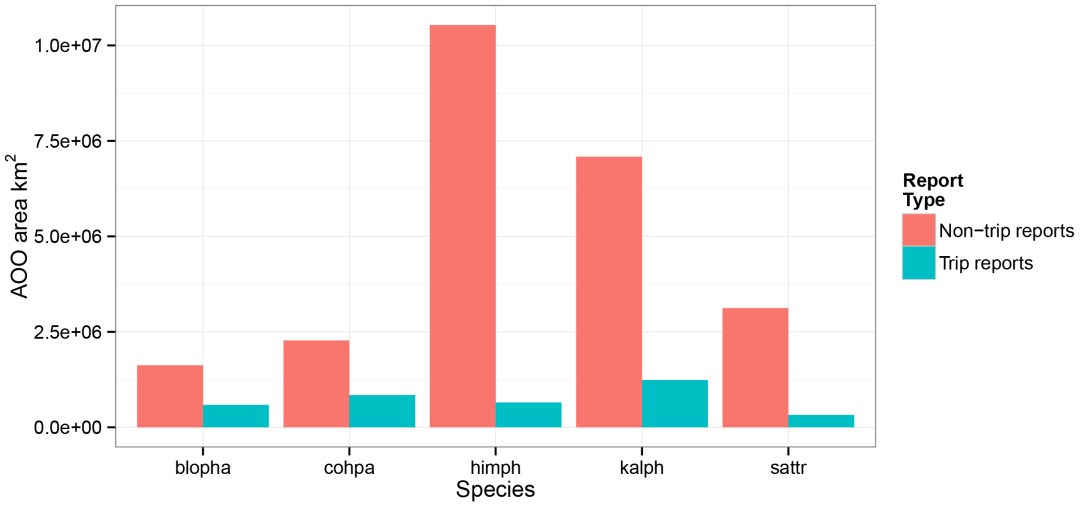


Figure A.16: Bar graph showing size of five species' ranges when constructed using trip reports vs. all other data sources. A paired t-test data revealed that ranges constructed from trip reports alone are significantly smaller than those constructed using other types of data (t = 5.04 df = 4, p = 0.01). Key: blopha = blood pheasant, cohpa = hill partridge, himph = Himalayan monal, kalph = kalij pheasant and sattr = satyr tragopan.

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